



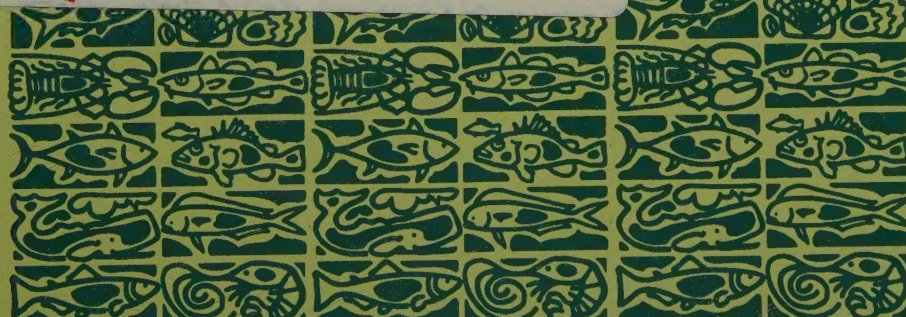
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Abstract—Because of a lack of fishery-dependent data, assessment of the recovery of fish stocks that undergo the most aggressive form of management, namely harvest moratoriums, remains a challenge. Large schools of red drum (*Sciaenops ocellatus*) were common along the northern Gulf of Mexico until the late 1980s when increased fishing effort quickly depleted the stock. After 24 years of harvest moratorium on red drum in federal waters, the stock is in need of reassessment; however, fishery-dependent data are not available in federal waters and fishery-independent data are limited. We document the distribution, age composition, growth, and condition of red drum in coastal waters of the north central Gulf of Mexico, using data collected from a nearshore, randomized, bottom longline survey. Age composition of the fishery-independent catch indicates low mortality of fish age 6 and above and confirms the effectiveness of the federal fishing moratorium. Bottom longline surveys may be a cost-effective method for developing fishery-independent indices for red drum provided additional effort can be added to nearshore waters (<20 m depth). As with most stocks under harvest bans, effective monitoring of the recovery of red drum will require the development of fishery-independent indices. With limited economic incentive to evaluate non-exploited stocks, the most cost-effective approach to developing such monitoring is expansion of existing fishery-independent surveys. We examine this possibility for red drum in the Gulf of Mexico and recommend the bottom longline survey conducted by the National Marine Fisheries Service expand effort in nearshore areas to allow for the development of long-term abundance indices for red drum.

Age composition and distribution of red drum (*Sciaenops ocellatus*) in offshore waters of the north central Gulf of Mexico: an evaluation of a stock under a federal harvest moratorium

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Data generated from the fishing industry are important for making population estimates and for evaluating management strategies in fisheries stock assessments. When stocks are severely depleted, harvest closures must be considered. Although such strong measures are likely to accomplish the goal of halting overfishing, the harvest ban simultaneously makes the determination of stock biomass status (e.g., overfished, recovered) increasingly difficult to assess. Although complete harvest bans are seldom used in the United States, recent mandates to end overfishing (2006 reauthorization of Magnuson-Stevens Fishery Conservation and Management Act [MSFCMA, 2006]) have required regional fisheries management councils and the National Marine Fisheries Service (NMFS) to consider this option more often. In 1987, red drum (*Sciaenops ocellatus*) in the Gulf of Mexico (GOM) was one of the first species to be designated as overfished; consequently, a fishery management plan (FMP) was developed and aggressive management actions were taken. As part of the GOM red drum FMP, the harvest of red drum in federal waters of the GOM was prohibited. After eighteen years of restrictions, red drum were no longer undergoing overfishing;

however, the stock condition remains unknown (listed as undefined in 2004; Hogarth¹). To properly evaluate this stock, updated biological information about adult red drum in the GOM is required. Typically, this information is collected directly from commercial and recreational fishermen, but owing to the harvest ban little is known about the current age structure, condition, and distribution of adult red drum in the GOM. We address these deficiencies in data and evaluate the current status of the red drum population in the north central Gulf of Mexico (ncGOM) and, most importantly, recommend monitoring measures for the stock.

Red drum occur throughout the GOM and along the Atlantic states to Massachusetts (Murphy and Taylor, 1990) and are the target of an important recreational fishery in all South Atlantic and GOM state waters. Red drum in the GOM use estuarine habitats as juveniles, including marshes, oyster reefs, seagrasses, and small creeks (Wenner, 1992; Rooker et al.,

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¹ Hogarth, W. T. 2004. A message from the NOAA assistant administrator for fisheries: Welcome to NOAA's National Marine Fisheries Service report on the status of the U.S. fisheries for 2004. NOAA, Washington D.C.

1998; Stunz et al., 1999; Stunz and Minello, 2001) and offshore habitats as adults (Beckman et al., 1988). Red drum reach adulthood and emigrate from the estuaries into offshore waters, typically between 3 and 6 years of age (Murphy and Taylor, 1990). Once offshore, these fish are believed to inhabit the waters along the continental shelf during most of the year. During the fall, adult red drum are known to aggregate near inlets to spawn (Overstreet, 1983). Spawning occurs along the continental shelf and within some estuary complexes, indicating that there is plasticity in spawning locations (Holt et al., 1985). Although not characterized as a coastal pelagic, adult red drum can be highly mobile; red drum have been recaptured farther than 700 km from where they were originally tagged (Overstreet, 1983). Because of this capacity for large-scale movement, management of red drum is dependent on cooperation among all GOM states and the federal government.

Harvest of red drum by both the commercial and recreational sectors has seen a marked change since the implementation of the FMP in 1987. Before the 1980s, the harvest was primarily commercial, with catches typically around 1–2 million kg yr⁻¹. By the mid-1980s, and the marketing of “blackened redfish,” the harvest increased to a maximum of 6 million kg yr⁻¹. The increased harvest occurred mostly in the offshore waters of the GOM and coincided with the development of a purse-seine fishery for adult red drum. Owing to the schooling behavior of red drum and the tendency of large schools to remain near the surface, red drum were easily targeted with the use of aerial spotters that direct purse-seine vessels. This expedited reduction in the number of adult fish led to decreased spawning stock biomass and resulted in the implementation of a FMP that required a total cessation of red drum harvest in federal waters in 1987. Since that time, commercial red drum harvest has remained near or below 40,000 kg yr⁻¹ (a small commercial fishery, however, exists in Mississippi state waters).

The recreational harvest of red drum is limited to state waters and remains heavily regulated. In contrast to the offshore commercial fishery of the 1980s, the historic and current recreational fishery targets primarily juveniles (1–4 yr old) in inshore habitats (Murphy and Crabtree, 2001). Annual landings have increased to over 6 million kg yr⁻¹. Even at this level, escapement of juveniles into the adult stocks was reported to be in excess of the 30% as required by the FMP (Powers and Burns²) for all GOM states. This increase suggests that the stock may be on a trajectory for full recovery; however, there is little current information about the

age, growth, condition, and population status of red drum that inhabit offshore waters, to evaluate this conclusion. To address this deficiency, we examined age composition, distribution, and condition of red drum from offshore waters in the ncGOM, using red drum collected during fishery-independent surveys from 2006 through 2010.

Materials and methods

Sample collection

Red drum (>660 mm total length, TL) were collected from Alabama, Mississippi, and federal offshore waters in the ncGOM during a monthly bottom longline survey ($n=428$ red drum, May 2006–May 2010). All longline set locations were randomly generated in predefined strata within the study area. From May 2006 through November 2008, sampling was stratified in blocks established along the shelf (east to west) as well as across the shelf (north to south). In both instances, sampling occurred from the shoreline (~2 m depth) to approximately the 20-meter isobath. Twelve stations were selected each month, allocated evenly across blocks and across the 2–20 m depth. Effort varied during the sampling period, with 93, 148, and 141 stations sampled in 2006, 2007, and 2008, respectively. Beginning in 2009, nearshore sampling was complemented with transect sampling wherein a line of longitude between 88°30′ and 87°30′W (the approximate longitudinal boundaries of Alabama) was randomly selected. Once chosen, stations were fished from the shoreline to approximately 200-m depth. Eighty nine stations were fished in 2009 and 30 in 2010 (Fig. 1A). For each longline set, commercial-style bottom longline gear was used. A monofilament mainline (454-kg test, 2-km length) was deployed off the stern through a block. High flier buoys were used at the start and end of each set. Five-kg weights (start, mid set, end set), and 3.66-m gangions (318-kg test) with 15/0 circle hooks were clipped to the mainline during deployment. Soak time was determined from the time the last high flier buoy was deployed until the first high flier buoy was retrieved to begin the haulback. Hooks were baited with Atlantic mackerel (*Scomber scombrus*) cut to fit the circle hooks (Driggers et al., 2008; Drymon et al., 2010). Measurements of abiotic variables were collected at each station with a Seabird SBE911-plus or an SBE25 conductivity-temperature-depth (CTD) probe (Sea-Bird Electronics, Inc., Bellevue, WA³). Nominal catch per unit of effort (CPUE) of red drum caught on the monthly longline survey was calculated as red drum · 100 hooks⁻¹ hour⁻¹. To standardize CPUE, the delta-lognormal index of relative abundance (I_y) as described by Lo et al. (1992) and Ingram et al. (2010) was estimated as

² Powers, S. P., and K. Burns. 2010. Summary report of the red drum special working group for the Gulf of Mexico Fishery Management Council. Special red drum assessment report. Prepared for the Gulf of Mexico Fishery Management Council, Scientific and Statistical Committee. [Available from <http://gulfcouncil.org/Beta/GMFMWeb/downloads/BB%20AUGUST%202010/G%20-%204%20Report%20of%20the%20Red%20Drum%20Working%20Group.pdf>, accessed March, 2012.]

³ Mention of trade names or commercial companies is for identification purposes only and does not imply endorsement by the National Marine Fisheries Service, NOAA.

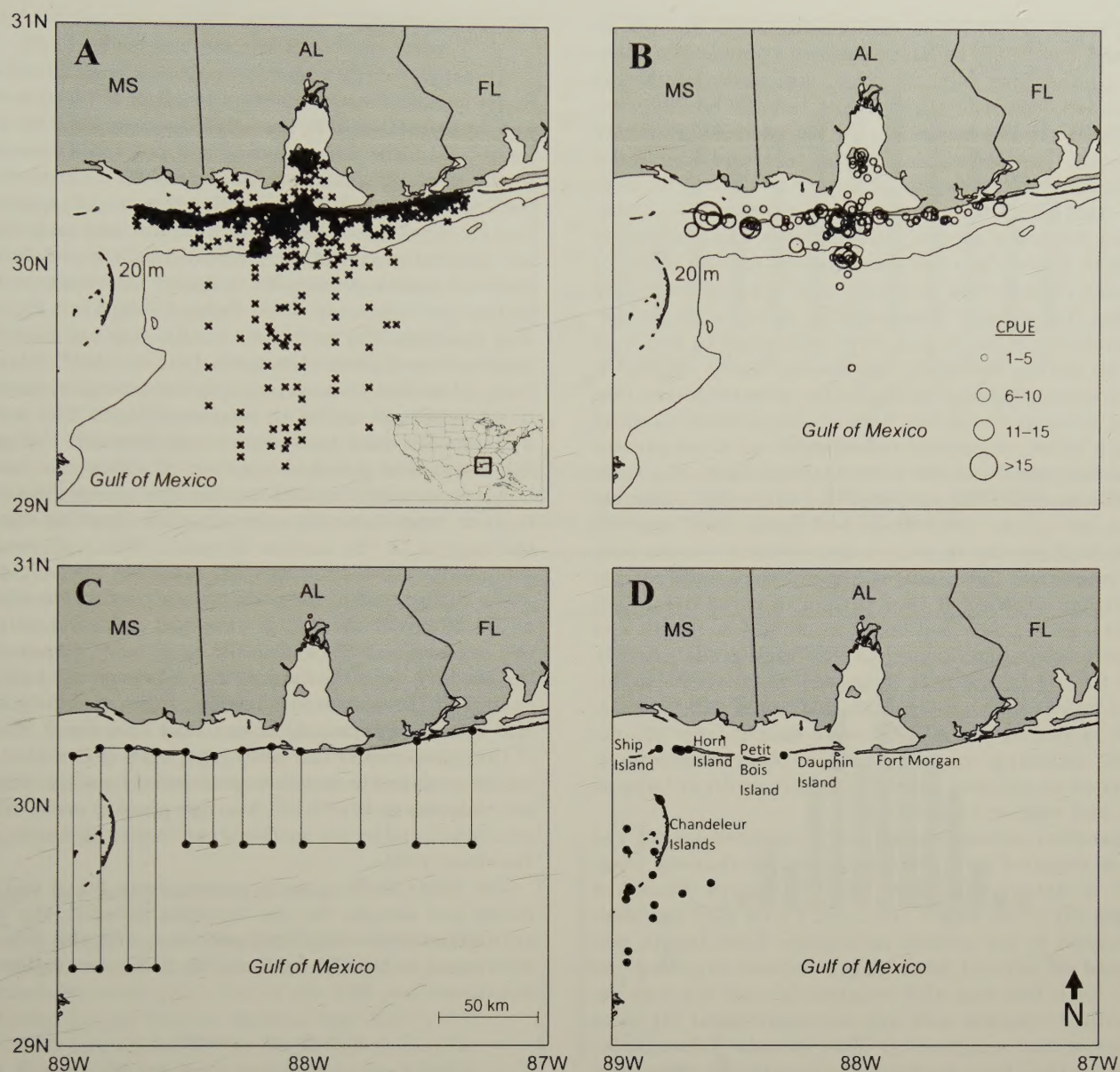


Figure 1

(A) Sampling locations and (B) catch per unit of effort (fish·100 hooks⁻¹ hour⁻¹) for red drum (*Sciaenops ocellatus*) collected between 2006 and 2010 during the bottom longline surveys. (C) Transect lines and (D) locations of sightings of red drum from the aerial surveys.

$$I_y = c_y p_y, \quad (1)$$

where c_y = the estimate of mean CPUE for positive catches only for year y ; and

p_y = the estimate of mean probability of occurrence during year y .

Both c_y and p_y were estimated with generalized linear models (GLMs). Data used to estimate abundance for positive catches (c) and probability of occurrence (p) were assumed to have lognormal and binomial distributions, respectively. The final standardized index is the product of the back-transformed year effects from

the two above mentioned GLMs. All GLMs were computed with year and month as factors. The standard error and coefficient of variation were estimated with a jackknife routine on factors with greater than two positive observations. Models were run by using the R programming environment, vers. 2.10.1 (R Development Core Team, 2009).

We complemented the red drum distribution data collected by bottom longline with data from aerial survey sightings for red drum conducted from August 2008 through March 2010. Aerial transect surveys were conducted with a Piper Apache PA-23 aircraft. The design covered the ncGOM from the shoreline to 20

nmi south (29°51'N) on eastern transects and out to 60 nmi south (29°19'N) on western transects to cover the Chandeleur Islands. The longitudinal coverage extended from the Chandeleur Islands in the west (88°59'W) to Pensacola, FL, in the east (87°19'W) (13 surveys). Two observers wearing polarized sunglasses conducted the survey by scanning the water to identify organisms from an average altitude of ~213 m. When fish were spotted, the observer signaled to the pilot to circle the object to confirm species identification, record GPS location, and take photographs of the fish school. These aerial surveys lasted approximately 8 hours and were conducted in two 4-hr flights with a fuel stop in between flights (Fig. 1C).

To increase the sample size for growth determination, we augmented our fishery-independent samples with a fishery-dependent collection of red drum greater than 660 mm TL at the 2009 Alabama Deep Sea Fishing Rodeo (ADSFR) on Dauphin Island, AL. Alabama state law allows 1 red drum >26 inches (660 mm) TL to be kept per day in state waters (within 3 miles from the coastline). Tournament anglers were entered into a random drawing if they turned in a red drum >26 inches. Prizes were not based on weight or length and hence anglers had no incentive to "high grade" the fish and turned in the first large red drum they caught. This tournament provided an additional 176 samples. Fish landed at the ADSFR were caught from the nc-GOM, including adjacent bays and coastal waterways, bounded on the east at longitude 85°, south at latitude 28°, and west at longitude 91°.

Standard measurements were collected for all red drum sampled by fishery-independent (bottom longline) or fishery-dependent (ADSFR) gears. Standard length (SL), fork length (FL), and TL for each fish were measured to the nearest millimeter. Total length was defined as natural total length without pinching the tail. Each fish was also weighed (g) and sex was determined. Gonads and any intraperitoneal fat were removed and weighed (g). Two-sample Kolmogorov-Smirnov (KS) tests were used to examine for differences in size distribution between fishery-independent and fishery-dependent sampled red drum.

Fish condition

Beginning in 2008, 3 indices were calculated to assess fish condition: gonadosomatic (GSI), intraperitoneal fat (IPF), and Fulton condition indices. Gonadosomatic and IPF indices are measures of condition where gonad weight (GSI) or fat content (IPF) is divided by total body weight then multiplied by 100 (Wilson and Nieland, 1994; Craig et al., 1995). The Fulton condition index was calculated by dividing weight of the fish by TL³ and multiplying by 100,000 (Ricker 1975). Differences in IPF between males and females were tested for using a nonparametric Mann-Whitney U test on pooled log(*x*+1) transformed data. Differences in sex ratio were tested for using a G-test (Zar, 1999) against an expected 1:1 male-to-female ratio.

Age determination

Beginning in 2008, ages were determined for all red drum captured on the bottom longline survey (*n*=227) and for fish collected at the ADSFR in July 2009 (*n*=176). Sagittal otoliths were removed and processed according to the methods described in the otolith manual of the Gulf States Marine Fisheries Commission (VanderKoooy and Guidon-Tisdell, 2003). Each otolith was weighed to the nearest gram. Material from the left otolith was removed starting from the anterior side with a thin-section saw (Hillquist, Inc., Denver, CO) until the core was reached. The sectioned otolith was polished and mounted on a glass slide with Loctite 349™ (Henkel Corp., Düsseldorf, Germany) light-sensitive glue and left to set overnight under an ultraviolet light. The otolith was then sectioned to approximately 0.50 mm. Each otolith section was polished and covered with a thin coat of liquid cover slip to smooth out any remaining scratches. Opaque zones (annuli) were counted from the core to the margin in the medial direction. The right otolith was used when the left was not available or when there was a disagreement between otolith readers (Beckman et al., 1988). Each otolith was aged independently by two readers, and the estimated ages were compared. If the reader's initial estimates did not agree, they jointly examined the otolith in question. If the resulting ages still disagreed, the otolith was read by a third reader. If the third reader did not agree with one of the two initial readers, the otolith was excluded from the sample set (Johnson et al., 2010). Average percent error (APE) was calculated by the methods outlined in Beamish and Fournier (1981).

Von Bertalanffy growth curves were fitted to both males and females for the complete data set, the fishery-independent (longline) data set, and the fishery-dependent (ADSFR) data set by using the following equation:

$$L_t = L_{\infty} [1 - e^{k(t-t_0)}], \quad (2)$$

where L_t = TL at time t ;

L_{∞} = the asymptotic length;

e = the base of natural logarithms;

k = the von Bertalanffy growth coefficient,

t = age; and

t_0 = the theoretical age at which TL equals zero (von Bertalanffy, 1938).

Differences in growth curves between males and females were tested for using a likelihood ratio test (Kimura, 1980; Haddon, 2000).

Results

Distribution of red drum

A total of 428 adult red drum were captured on bottom longline cruises at multiple locations from Mississippi

and Alabama from 2006 through 2010 (Fig. 1B). The highest concentrations of fish were collected near the pass between Dauphin Island and Ft. Morgan (mouth of Mobile Bay) and offshore Petit Bois Island. Twenty-four percent of sets produced red drum, and a maximum of 23 individuals were captured during a single set (October 2007). Standardized mean monthly CPUE for red drum was bimodal with the greatest concentrations of fish captured in March ($2.7 \text{ fish} \cdot 100 \text{ hooks}^{-1} \text{ hour}^{-1}$) and April ($2.3 \text{ fish} \cdot 100 \text{ hooks}^{-1} \text{ hour}^{-1}$) followed by November ($1.5 \text{ fish} \cdot 100 \text{ hooks}^{-1} \text{ hour}^{-1}$). Mean monthly temperatures followed a predictable seasonal pattern, ranging from a low of 12.3°C in February to 31.7°C in August (Fig. 2). The majority of red drum were collected in water $<20 \text{ m}$ deep, although effort was high throughout waters $<60 \text{ m}$ deep. The maximum depth red drum were collected from was 63 m .

In general, red drum distributions revealed in the aerial surveys complemented the spatial distribution patterns determined from the bottom longline gear. Red drum schools were spotted during 8 out of 13 aerial surveys. All red drum schools were spotted west of Mobile Bay (88°W longitude), many near the barrier islands of the nGOM. Sightings occurred most frequently in the shallow waters around Dauphin Island and Petit Bois Island, and around the Chandeleur Islands (Fig. 1D).

Length, age, and growth of red drum

Length-frequency distributions differed between the longline- and ADSFR-collected fish. The mean ($\pm$ standard error [SE] TL) of the 428 red drum collected on the bottom longline was $912 (\pm 3.0) \text{ mm}$ with a range of 720 mm to 1101 mm TL (Fig. 3A). For the 176 red drum collected at the ADSFR, the mean total length was 849 mm with a range between 660 and 1156 mm TL (Fig. 3B). Fish collected on longlines were significantly longer than the fish captured at the ADSFR (two-sample KS test, $D=0.416$, $P<0.0001$). Males and females were analyzed separately for length differences. Females were longer than males for both bottom longlines and ADSFR. Mean total length for bottom longline females ($n=114$) was $923 (\pm 6.09)$ and $885 (\pm 6.23)$ for males ($n=99$). For the ADSFR, mean total length for females ($n=106$) was $870 (\pm 12.25)$ and $811 (\pm 15.03)$ for males ($n=67$).

Ages were determined for both fishery-independent and fishery-dependent collections. Average percent error for the two readers was 0.004% , indicating precise aging. No otoliths were excluded owing to a discrepancy in counts of annuli. The mean ($\pm$ SE) age for fish caught on the longline was $16.5 (\pm 0.4)$ years with a range of 2–34 years (Fig. 4A). For fish sampled

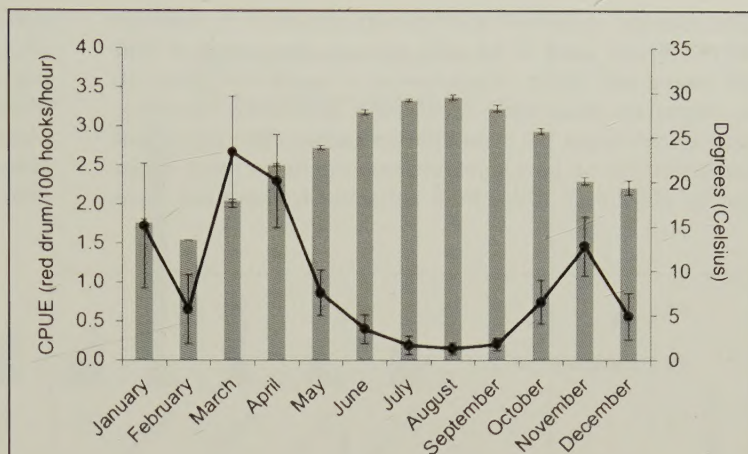


Figure 2

Mean monthly catch per unit of effort (CPUE $\pm$ standard error, SE), line) and temperature ($\pm$ SE, bar) for longline-caught red drum (*Sciaenops ocellatus*), May 2006–May 2010.

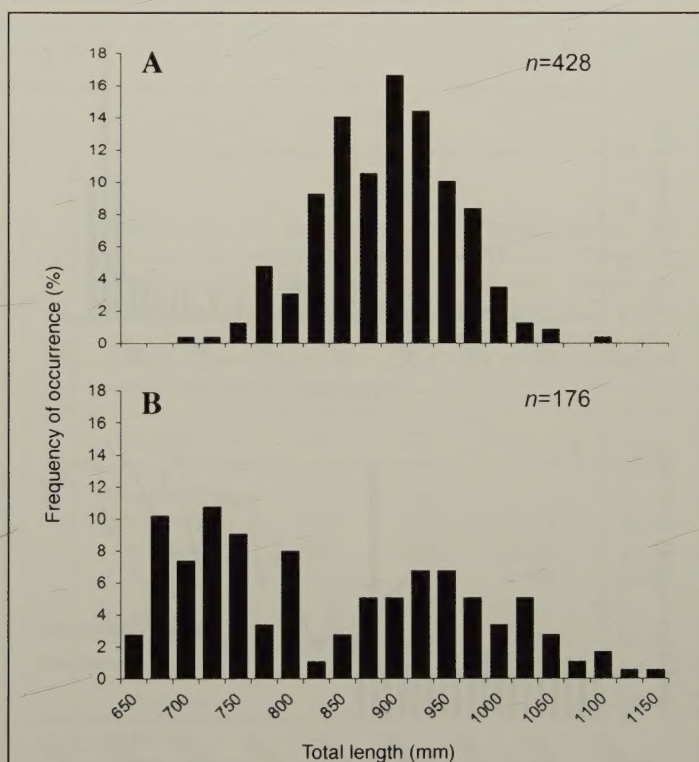


Figure 3

(A) Length frequency of red drum (*Sciaenops ocellatus*) sampled on the fisheries-independent longlines (2006–10) and (B) at the fisheries-dependent Alabama Deep Sea Fishing Rodeo (2009).

at the ADSFR, age composition ranged from 2 to 38 years. The mean age of an ADSFR fish was 9 years; however, 50% of fish collected were in the 2–4 year category (Fig. 4B). For longline fish, the age frequency

distribution indicated a relatively constant proportion of red drum ages 5–24 and a lower frequency of fish 25 years and older. Assignment of aged red drum to a respective year class revealed a decreased contribution of red drum born in 1986 or earlier. The increased contribution of postmoratorium red drum was highest in 1992 and 1993. Fish collected by longline were

significantly older than fish collected from the ADSFR ($F_{2,643}=51.82$, $P<0.001$). For the ADSFR-collected fish, higher proportions of 2-, 3-, and 4-year-old fish were evident in the distribution, alongside a relatively constant frequency of 7–24 year-old fish. The recreational fishery for red drum >660 mm was dominated by recent year classes (Fig. 4).

The von Bertalanffy growth model parameter estimates indicated disparities between males and females, similar to those seen in previous studies (Table 1). We determined the sex for a total of 387 fish from both longlines and the ADSFR and calculated a 1:1.3 male-to-female ratio. For the fish collected in this study, the population had a significant female bias ($G_{\text{obs}}=3.932$, $df=1$, $P<0.05$). Higher L_{∞} were modeled for female red drum (1012 mm TL) than for males (969 mm TL), and females showed lower growth coefficients. This pattern was also seen with the data from each collection separately: bottom longline ($L_{\infty \text{ females}}=989$ mm TL, $L_{\infty \text{ males}}=954$ mm TL) and ADSFR ($L_{\infty \text{ females}}=1046$ mm TL, $L_{\infty \text{ males}}=1009$ mm TL) (Table 2).

Condition indices

The Fulton condition index remained relatively constant by month and sex, whereas the GSI and IPF indices varied throughout the year and between sexes. GSI values were highest in September and relatively low for nonsummer months. IPF peaked in May, remained relatively consistent through June and July and abruptly decreased in September. IPF remained low through the November sampling (Fig. 5). Males (mean IPF [$\pm$ SE]=0.55 [$\pm$ 0.06]) had significantly less IPF compared to females (mean IPF [$\pm$ SE]=0.65 [$\pm$ 0.07]) (Mann-Whitney U test, $Z=-3.233$, $P=0.001$).

Discussion

The mandate to end overfishing included in the Magnuson-Stevens Fisheries Conservation and Management Act (2006) requires aggressive management for species subjected to unsustainable exploitation levels. Harvest moratoriums may be viewed as the extreme end of a continuum of potential management interventions. Although full harvest closures are rare, harvest closures of any kind will decrease the quantity of fisheries-dependent data available for stock assessments. For species whose harvest moratoriums persist for extended periods or over large spatial scales, the lack of fisheries-dependent data may severely limit the ability to assess stock condition. Red drum are an example of such a scenario. The total ban on harvest in federal waters in the Gulf of Mexico has been in place since 1987 and little information is available on the spawning stock of

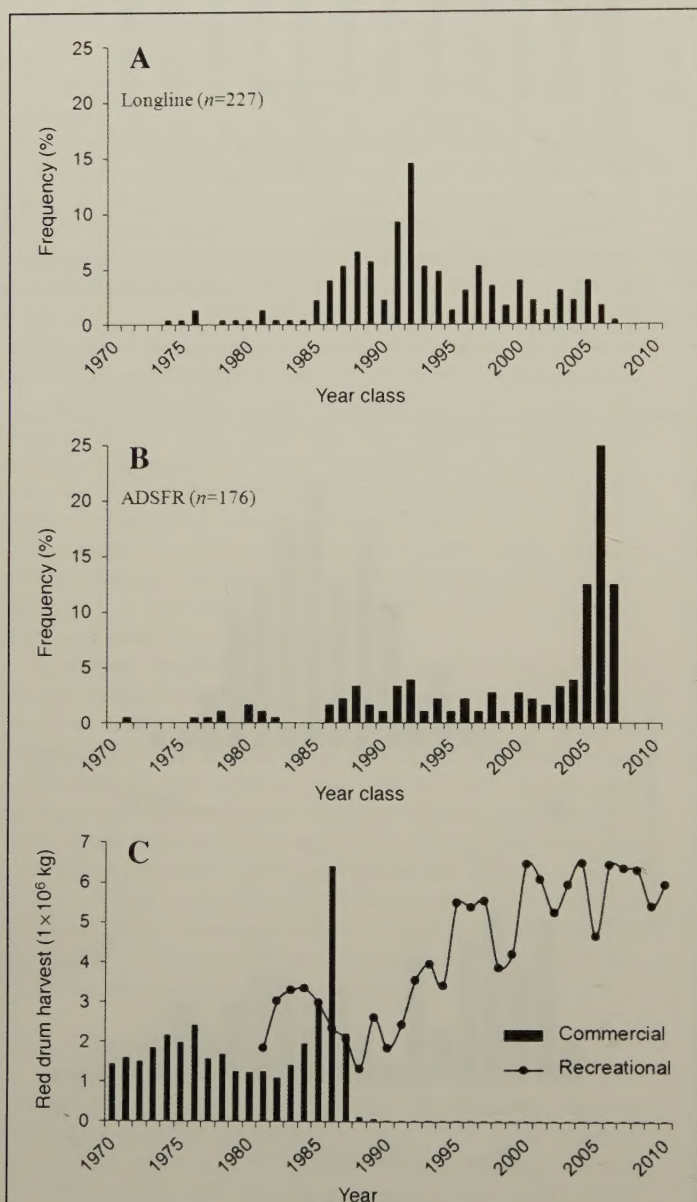


Figure 4

Age composition of red drum (*Sciaenops ocellatus*) sampled on the (A) fisheries-independent longlines (2006–10), and (B) at the fisheries-dependent Alabama Deep Sea Fishing Rodeo (ADSFR) (2009). (C) Commercial and recreational harvest of red drum from the Gulf of Mexico. Commercial data are available from <http://www.st.nmfs.noaa.gov/st1/commercial>, and recreational data are available from <http://www.st.nmfs.noaa.gov/st1/recreational> (accessed March 2012).

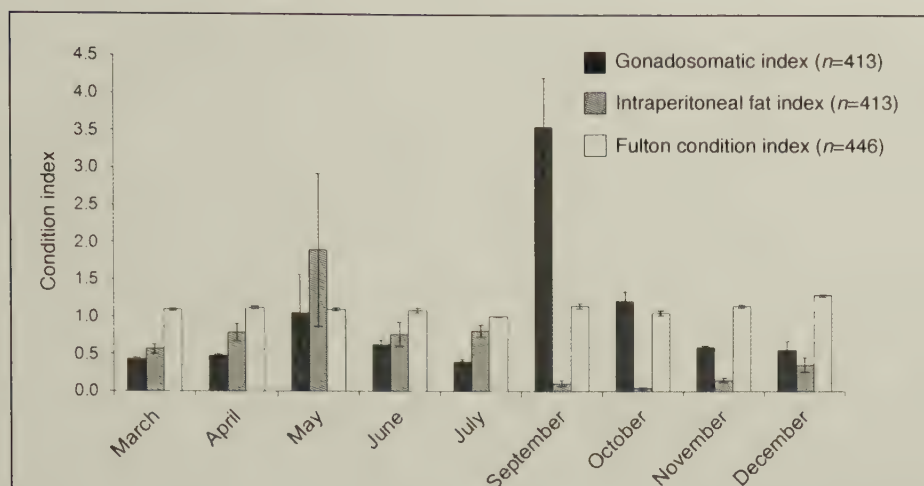


Figure 5

Mean monthly gonadosomatic index (GSI), intraperitoneal fat index (IPF), and Fulton condition indices for red drum (*Sciaenops ocellatus*) sampled from the north central Gulf of Mexico between March 2008 and May 2010. Gonadosomatic and IPF indices are measures of condition where gonad weight (GSI) or fat content (IPF) is divided by total body weight then multiplied by 100 (Wilson and Nieland, 1994; Craig et al., 1995). The Fulton condition index was calculated by dividing weight of the fish by TL^3 and multiplying by 100,000 (Ricker, 1975).

Table 1

Comparison of von Bertalanffy growth parameter estimates from red drum (*Sciaenops ocellatus*) studies in the Gulf of Mexico. L_{∞} =the asymptotic length, k =the von Bertalanffy growth coefficient, and t_0 =the theoretical age at which total length equals zero. TX=Texas; LA=Louisiana; MS=Mississippi; AL=Alabama; FL=Florida.

Study	Location	L_{∞}	k	t_0
Current study	AL			
Males		923	0.11	-10.00
Females		965	0.109	-10.00
Beckman et al. (1988)	TX, LA, MS, AL			
Males		909	0.137	-7.74
Females		1013	0.088	-11.29
Murphy and Taylor (1990)	FL	934	0.45	0.029

this species. Effective monitoring of the recovery of red drum stocks will require the development of fishery-independent indices that allow periodic examination of age composition—a strategy needed for many species under harvest moratorium. The most cost-effective approach to developing such monitoring is the expansion of existing fishery-independent surveys because of the limited economic incentive to evaluate nonexploited stocks. Using our bottom longline survey, which was designed to be an expansion of the NMFS Gulf of

Table 2

Parameters for von Bertalanffy growth function by survey type (combined, longline and Alabama Deep Sea Fishing Rodeo [ADSFR]) and sex for red drum (*Sciaenops ocellatus*) in the north central Gulf of Mexico. N =total sample size, L_{∞} =the asymptotic length, k =the von Bertalanffy growth coefficient, and t_0 =the theoretical age at which total length equals zero.

Data set	N	L_{∞}	k	t_0
Combined				
(both sexes)	403	993	0.109	-10.00
Males	166	969	0.110	-10.00
Females	221	1012	0.109	-10.00
Longline				
(both sexes)	227	979	0.108	-10.00
Males	99	954	0.114	-10.00
Females	114	989	0.113	-10.00
ADSFR				
(both sexes)	176	1037	0.127	-7.11
Males	67	1009	0.123	-7.58
Females	106	1046	0.126	-7.36

Mexico bottom longline survey into nearshore waters, we illustrate the benefits of such an approach.

Fishery-independent longline surveys proved to be an effective means of obtaining the data needed to calculate an abundance index for adult red drum in the nearshore waters of the ncGOM. Data from these

surveys, coupled with fishery-dependent sampling in state waters, provide a means by which to calculate age composition, length frequency, and condition of ncGOM red drum. This combination of fishery-dependent and fishery-independent techniques allows an assessment of a population whose spawning stock is largely under a harvest moratorium.

Monthly red drum CPUE showed a bimodal distribution with peak catch rates occurring in spring (March to April) and fall (November). These peaks in catch may reflect the preferred temperature range of this species. Red drum CPUE dropped precipitously from May through September, during which time mean monthly water temperature averaged 28°C (± 0.18 SE). During the spring (March–April) and fall (November) peaks in CPUE, mean bottom temperature was 21 (± 0.31 SE) and 20 (± 0.39 SE) °C, respectively. These data indicate that the distribution of red drum in coastal waters may be temperature dependent. Under laboratory conditions, greatest metabolic capacity was shown for red drum when exposed to near-optimum thermal regimes (Fontaine et al., 2007). Equally plausible, seasonal patterns in red drum CPUE may be linked to temporal changes in prey availability, as demonstrated for red drum in Texas (Scharf and Schlicht, 2000), although we lack the data to further examine this hypothesis. Regardless of the mechanism, the bimodal peak in distribution determined with the current sampling scheme has implications for designing monitoring programs. Our data indicate the benefit of monthly sampling when possible and provide evidence that surveys sampling exclusively during summer months will not provide high enough catches to generate specimens for age determination.

In contrast to the utility of bottom longlines, high variability in water clarity and sea conditions limited data from the aerial surveys. The aerial surveys were most useful in providing spatial distribution data. Ideally, abundances of red drum could be calculated from the aerial survey images; however, these abundances would only represent fish on the surface and not those found at depth within the school. Also, during the aerial survey, high variability in water clarity and sea state made it difficult to enumerate fish. Therefore, number of schools was the best descriptor of abundance. School size (area coverage) could be calculated from images where the lens size and altitude were known; however, lens angle is a significant covariate that is difficult to control. Logistic difficulties with aerial surveys and the fact that the survey produced no length or age estimates lead us to question the utility of these surveys in generating long term abundance indices.

Age composition of the red drum collected by fishery-independent sampling indicated that older age classes of red drum are present in the offshore population of Alabama and Mississippi. Red drum younger than 24 years old were present in much higher frequency than 25+ year-old fish. The age frequency distributions of 6–24 year-old fish was relatively constant with an average of 4.0 ($\pm 1.3\%$, 1 standard error) in each age class. In con-

trast, fish older than 26 years were present in low frequencies (0.7 [$\pm 0.4\%$]) per age class. Fish 25 years old at time of capture would be assigned to annual cohorts between 1983 and 1985 and therefore subject to fishing pressure prior to the federal moratorium (1987). Our findings agree with previous work conducted in Florida, where the distribution of adult red drum sampled with purse seines showed that fish before the 1984 year class were rare (Murphy and Crabtree, 2001).

Further evidence for the effectiveness of the moratorium was provided by the elevated frequencies of the 1991 and 1992 year classes. Red drum recruit to the offshore population between the ages of 3 and 6; therefore, the year classes in 1991 and 1992 would have been the first to recruit to the offshore population after the 1987 federal fishing moratorium. Data for the fish in year classes from 1987 through 1992 therefore provide evidence that the moratorium was effective in increasing recruitment to the offshore fishery. Using similar techniques, Murphy and Crabtree (2001) reported strong red drum year classes in 1986 and 1989. The authors attribute the persistence of these year classes to periods of low exploitation after state management actions. These studies show how the effectiveness of management measures can be tested by fishery-independent monitoring of year-class strength.

Our fishery-independent collections also allowed evaluation of individual fish condition and provide insight into timing of red drum spawning. Condition indices for longline-collected fish were relatively high, with the exception of the period after spawning. Our data suggest that those energy reserves stored as intraperitoneal fat are consumed during the energetically demanding red drum spawning period. In a concurrent survey of ichthyoplankton (January–December, 2006–10), Hernandez et al. (2010) reported red drum larvae only in September and October. This period of larval occurrence coincides with the highest GSI and lowest IPF indices and is consistent with a late September to early October spawning period.

Although a contrast between the fishery-independent and fishery-dependent collections is limited because the ADSFR collection was not a random sampling of all anglers in the region, contrasting the fisheries independent and dependent collections does provide a snapshot of the data that would be generated from future recreational surveys of red drum above the slot limit (i.e., fish within an allowable minimum and maximum length for harvest, 16–26 inches in AL). The comparison highlights potential differences and biases that are germane to monitoring a stock under the current federal harvest moratorium. The offshore population of red drum that serves as the spawning stock straddles the boundary between state and federal waters south of Alabama, Mississippi, and Louisiana (three miles from shore). From interviews with the 176 anglers that provided fish to the ADSFR, most anglers caught red drum in the higher salinity areas of Mobile Bay and Mississippi Sound and areas within a mile from shore

(e.g., Dixie Bar, AL). Less than 25% of anglers reported fishing in waters greater than 3 miles offshore near artificial reefs. The majority (60%) of our bottom longline sampling effort occurred in state waters with the remaining 40% in federal waters. The vast majority of our catch occurred within 3 miles from shore and was highest near the inlets. Given that the spatial distribution of effort between the fishery-dependent and independent collections overlaps, differences between size and age frequency between the fishery-dependent and independent collections are likely due to the broader selectivity of the recreational (hook and line) gear than to the standardized hook size and bait type in our bottom longline survey. The bottom longline selected for fish age 6 and above (>800 mm), whereas 50% of the hook and line catch was 2–5 year-old fish and did not provide large sample sizes of older age classes. The bottom longline tended to capture older age classes of red drum and their abundance is a key determinant for stock condition for long-lived fish.

Central to the current management scheme for red drum is protection of the spawning population. A post-moratorium increase in adult red drum is apparent from examination of the year-class frequency in our data; however, the large increase after the moratorium declined to more modest levels in the last ten years during which fish could be considered fully selected for the longline gear (1996–2006). This trend corresponds to the overall landings pattern for the GOM. Peak, and apparently unsustainable, commercial landings occurred in 1986 around 6.3 million kg. After the moratorium, total landings decreased to 1.5 million kg annually but have steadily increased and recreational landings were approximately 6.3 million kg in 2010. Currently, Louisiana, Mississippi, and Alabama allow retention of one red drum above the catch limit per day, whereas Texas allows two oversized red drum per year. Data collected from the ADSFR suggest 2–5 year-old fish are readily available to recreational anglers and the recreational fishery could be a significant source of mortality for the spawning stock. Although our data represent a limited geographic range of the stock, the data indicate the need to re-assess the current management scheme and evaluate the bag limits and escapement rates of red drum.

Conclusions

Assessment of stock conditions for fish populations under severe harvest restrictions is complicated by a lack of fishery-dependent data. Fishery-independent surveys could be used to fill this void; however, funds for fishery-independent surveys targeting nonexploited stocks are limited. Key to monitoring these stocks will be increasing the efficiency of data collection (*sensu* Link et al., 2008). Stock assessments of red drum are complicated by the mosaic of differing state regulations and the harvest ban in federal waters. Commercial catch from the GOM red drum fishery is minimal (Mis-

issippi allows a limited harvest) and recreational size limits vary between states. Given these differing spatial management schemes, fishery-independent sampling is critical for establishing long-term abundance indices and determining age composition across the GOM for effective monitoring. Our results suggest that bottom longline surveys could fill this role. Specifically, designs should be optimized by focusing in coastal waters <20 m deep, where 99% of our red drum catch occurs. Targeting effort in the spring (March–April) and fall (October–November) would be expected to maximize the sample size for age determination. Although annual bottom longline surveys are currently conducted by the National Marine Fisheries Service (*see* Driggers et al., 2008) and provide fishery-independent data for sharks and red snapper, effort is limited in <20 m depth and the survey occurs only from July through September. Expanding already existing fishery-independent surveys is likely the most cost-effective method of examining stocks that have limited fishery-dependent data.

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Abstract—Cannibalism is thought to be an influential top-down process affecting walleye pollock (*Theragra chalcogramma*) recruitment in the eastern Bering Sea (EBS). In summer, many age-1 pollock occupy the same depths as those of adult walleye pollock, making them vulnerable to cannibalism. We examine factors that influence the occurrence and amount of cannibalism, as well as the abundance and co-occurrence of predator and prey walleye pollock. Large walleye pollock were generally found in deeper waters and avoided cold temperatures; whereas, age-1 walleye pollock were found in broader bottom depth and temperature ranges. The occurrence of cannibalism was highest in the area where predator and prey walleye pollock co-occurred and the amount of cannibalism was highest on the middle and outer EBS shelf. Both the occurrence and amount of cannibalism were influenced by location, bottom temperature and bottom depth, and the abundance of prey walleye pollock. The abundance of both large and small walleye pollock decreased during the 1982–2006 survey period in the EBS and, hence, the occurrence and amount of cannibalism also decreased. The occurrence and amount of cannibalism observed in the diet samples from the summer survey were good indicators of year-class strength, as estimated by the stock assessment model. There was more cannibalism of age-1 walleye pollock when predicted recruit abundance was highest, indicating that summer cannibalism on age-1 walleye pollock, a top-down process, does not control walleye pollock recruitment in the EBS.

Factors influencing cannibalism and abundance of walleye pollock (*Theragra chalcogramma*) on the eastern Bering Sea shelf, 1982–2006

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Recruitment of fish to fisheries depends on survival during the early life history (Hjort, 1914; Parker, 1966; Bailey and Spring, 1992; Bradford, 1992). Factors that can affect survival during early life stages and subsequent recruitment to fisheries may be related to environmental or bottom-up factors (Hollowed and Wooster, 1992; Hunt et al., 2002) and predatory or top-down factors (Wespestad et al., 2000; Hunt et al., 2002; Mueter et al., 2006). Both top-down and bottom-up forces have been hypothesized as contributing to recruitment of one of the largest fishery resources in the eastern Bering Sea, walleye pollock (*Theragra chalcogramma*; hereafter referred to as pollock; Mueter et al., 2006).

Pollock in the eastern Bering Sea (EBS) comprise the highest-volume commercial fishery in the United States (NMFS, 2010). Their recruitment is widely believed to be influenced by a combination of environmental and density-dependent effects (Hollowed and Wooster, 1992; Mueter et al., 2006). In the EBS, for example, bottom-up forces that may affect pollock recruitment include the effect of the extent, timing, and duration of ice, ocean currents, and temperature on primary and secondary productivity (Wespestad et al., 2000; Mueter et

al. 2006; Coyle et al., 2011; Hunt et al., 2011). Top-down processes, such as predation, including cannibalism, are also important influences on EBS pollock survival (Dwyer et al., 1987; Mueter et al., 2006, 2011).

Cannibalism is recognized as an important factor occurring year-round and affecting the survival and recruitment of pollock in the EBS (Bailey and Houde, 1989; Mueter et al., 2006, 2011). In the fall, age-0 pollock are heavily cannibalized by older pollock (Dwyer et al., 1987), a density-dependent process that is hypothesized to affect the year-class strength and recruitment to the adult population (Wespestad et al., 2000). Additionally, in the summer, cannibalism on age-1 pollock occurs (Bailey and Dunn, 1979; Francis and Bailey, 1983; Dwyer et al., 1986, 1987) and may also affect subsequent year-class strength. Factors hypothesized to affect pollock cannibalism include: water column stratification and temperature and their effects on food availability; and the horizontal and vertical overlap of prey and predator pollock (Francis and Bailey, 1983; Bailey, 1989; Wespestad et al., 2000; Duffy-Anderson et al., 2003; Mueter et al., 2006; Hunt et al., 2011).

Summer temperatures and water column stratification on the EBS shelf

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are largely determined by sea-ice conditions during the preceding winter and mixing forces during the summer (Kachel et al., 2002; Stabenot et al., 2010). As a result, the EBS shelf comprises 3 depth domains (generally, coastal <50 m, middle 50–100 m, and outer 100–180 m), each with its own hydrographic features: a mixed water column in the coastal domain, a 2-layered water column in the middle domain, and a 3-layered water column in the outer domain (Stabenot et al., 2001). Water temperature and stratification can affect the amount of primary and secondary production, the growth and timing of zooplankton, and zooplankton species composition (Stabenot et al., 2001; Napp et al., 2002; Rho et al., 2005), thereby affecting food availability for EBS fish and their subsequent survival. Mueter et al.¹ found evidence for environmental effects on age-1 pollock survival and recruitment. During November to March, ice forms and extends from the Bering Strait in the north to the Alaska Peninsula in the south (Napp et al., 2000). The southerly extent and timing of ice (arrival and persistence) in winter and early spring have a strong effect on the size and southerly extent of the cold pool (bottom water <2°C) (Stabenot et al., 1998, 2001; Kachel et al., 2002). The cold pool can directly affect the distribution of some upper trophic level species, including adult pollock (Napp et al., 2000).

The vertical and horizontal distribution of pollock varies ontogenetically, seasonally, and temporally (Hinckley, 1987; Kotwicki et al., 2005; Bacheler et al., 2010). Adults generally spawn at depths between 100 and 250 m from January to August, depending on the spawning area (Hinckley, 1987; Bacheler et al., 2010). As part of a feeding migration in summer, most pollock on the shelf migrate towards the northwest and pollock on the southeast shelf migrate towards the northeast (Kotwicki et al., 2005). Adult pollock are demersal and distributed on the outer and middle depth domains in summer. Adults prefer cooler and stable temperatures (Duffy-Anderson et al., 2003) but avoid the cold pool (Chen, 1983; Bakkala and Alton, 1986). Upon hatching, young pollock larvae are distributed in the upper water column and are subject to currents and wind-driven advection (Nishiyama et al., 1986). Larvae undertake small diel vertical migrations within the upper 30 m of the water column (Pritchett and Halderson, 1988). Bailey (1989) observed that, in the fall months, small-size age-0 pollock (<60 mm fork length) remained in surface EBS waters above the thermocline, but larger age-0 pollock (70–89 mm fork length) moved to deeper depths, below the thermocline in the daytime, thereby making them vulnerable to cannibalistic adults at that time of year. In the summer months, older pollock (age-1 and age-2) have been found within and inshore of the cold

pool (Francis and Bailey, 1983), as well as throughout the water column (Bakkala and Alton, 1986) because they are able to tolerate more variable temperatures than adults (Chen, 1983; Duffy-Anderson et al., 2003). There is a tendency for age-1 and age-2 pollock to be vertically separated, with the age-1 pollock near the bottom and age-2 pollock distributed higher in the water column (Duffy-Anderson et al., 2003). In the summer months, therefore, age-1 pollock are vulnerable to cannibalistic adults. In fact, in the EBS during summer, age-1 pollock comprised the majority of cannibalized prey pollock (Dwyer et al., 1987). The factors that determine the occurrence and amount of cannibalism on age-1 pollock are not well understood.

Our goal was to determine the factors that affect the occurrence and amount of cannibalism on vulnerable pollock in the summer. Age-1 pollock often occupy the same depths as adult pollock in the summer, thereby making them vulnerable to cannibalism (Dwyer et al., 1987; Duffy-Anderson et al., 2003). Our specific objectives were to determine which environmental factors influence the occurrence and amount of pollock cannibalism, as well as the factors that influence the abundance of predator and prey pollock and their co-occurrence. One working hypothesis was that the abundance of large predatory pollock and small prey pollock and their co-occurrence are determined by temperature, depth, location, the presence of the cold pool, and year. These covariates were also examined for their influence on the occurrence of cannibalism, along with other covariates: the size of predators, and the co-occurrence of both large predatory pollock (>200 mm standard length [SL]) and small prey pollock. Owing to their vertical distribution in the water column, small prey pollock that may be vulnerable to cannibalism measure 60 to 200 mm SL, a size that corresponds to the size of age-1 pollock in the summer. A final hypothesis tested was that, where cannibalism occurs, the amount of cannibalism is determined by the size of predators, bottom temperature, bottom depth, location, the presence of the cold pool, year, and the abundance of vulnerable prey pollock (60–200 mm SL).

Materials and methods

Pollock abundance

Abundance, co-occurrence, and diet data on predator and prey pollock were collected during the annual National Marine Fisheries Service (NMFS) bottom trawl surveys of the EBS shelf (≤200 m depth) from 1982 to 2006 (excluding 1983 and 1984, when pollock diet data were not collected). Groundfish species were captured at stations on a regular grid of the EBS shelf (Fig. 1). The details of the survey design and sampling methods are described in Lauth (2010) and Stauffer (2004). Pollock captured in the bottom trawl were weighed and measured for length. A subset of pollock was selected for stomach content analyses. Pollock catch per unit of effort (CPUE,

¹ Mueter, F. J., M. C. Palmer, and B. L. Norcross. 2004. Environmental predictors of walleye pollock recruitment on the Eastern Bering Sea shelf. Pollock Conservation Cooperative Research Center, Univ. Alaska, Fairbanks. [Available from <http://www.sfos.uaf.edu/pcc/projects/03/norcross/MueterNorcrossFinalReport2003.pdf>, accessed April 2012.]

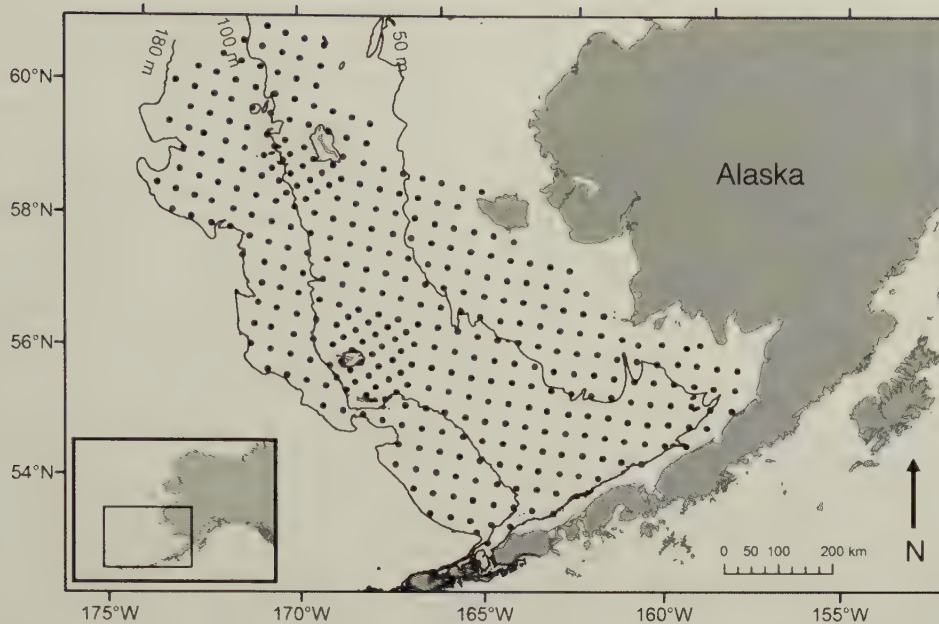


Figure 1

Study area on the eastern Bering Sea shelf (≤ 200 m depth) showing the stations sampled annually in the National Marine Fisheries Service bottom trawl survey for groundfish species, including walleye pollock (*Theragra chalcogramma*), during 1982–2006 (excluding 1983 and 1984). Three depth domains of the shelf are delineated by depth contours (coastal <50 m, inner 50–100 m, and outer 100–180 m).

number of fish per hectare) was estimated by size class by using the area swept by the net (average net width measured during the bottom trawl haul multiplied by the distance the net was towed over the seafloor). Based on diet information, the assumption was made that small pollock caught in the survey were potential prey (sum of small [60–200 mm SL] pollock CPUE; *SmallPollock*), and large pollock were assumed to be potential predators (sum of large [>200 mm SL] pollock CPUE; *LgPollock*). Predator-prey co-occurrence (*Overlap*) between *SmallPollock* and *LgPollock* was determined for each station and designated a value of 1 (co-occurrence) if the CPUE values of both were greater than zero, otherwise a value of 0 was assigned (no co-occurrence).

Bottom temperature, cold pool, and depth

Water temperature and depth profiles were collected at each station by using bathythermographs attached to the headrope of the bottom trawl net (expendable models before 1993 and microbathythermographs in later years). Bottom depths and temperatures were recorded for each station. It was also noted whether the cold pool (temperatures $<2^{\circ}\text{C}$) was present or absent at each station (binomial variable, *ColdPool*; see Buckley et al. [2009] for a description).

Pollock abundance models

General additive models (GAMs) were used to explore the relationship between pollock abundance and pre-

dictor variables (covariates). The 3 response variables examined were large predatory pollock CPUE (>200 mm SL, *LgPollock*), small prey pollock CPUE (60–200 mm SL, *SmallPollock*), and the co-occurrence of *LgPollock* and *SmallPollock* (*Overlap*). Pollock CPUE values (*SmallPollock* and *LgPollock*) were $\ln+1$ transformed to achieve normality. Because the timing of the survey and overall abundance of pollock changed from year to year, “Year” was included as a categorical predictor. Pollock catch or the occurrence of cannibalism (e.g., inshore vs. offshore stations) may have been influenced by station location; therefore, station location (latitude and longitude) was included as a single, smoothed bivariate term, $s(\text{latitude}, \text{longitude})$, and referred to as “Location” in the GAMs. Latitude may limit the northerly distribution of pollock and longitude may be important in terms of distribution with depth. Because bottom temperature and bottom depth are related, a smoothed bottom temperature and bottom depth interaction term was included (*TempDepth*). The presence or absence of the cold pool was included as a binomial term (*ColdPool*) in the model as well. Survey duration typically extended over the entire summer period from June to August (i.e., fish likely grew or moved during the time it took to complete the survey and collect the data in one year), therefore Julian day was considered for inclusion in models. The groundfish survey, however, began in the southeast and proceeded to the northwest Bering Sea, and the day of year was significantly correlated with latitude and longitude (coefficient of determination [r^2] = 0.390 and 0.817, $P < 0.001$ and < 0.001 , respectively).

Separation of the effect of Julian day from the effect of location on abundance and cannibalism was, therefore, not possible and Julian day was not directly included in further analyses.

A step-wise model-selection approach was used for GAMs, where all covariates were included in initial models and covariates with the least significant P -values ($P > 0.05$) were removed one at a time in subsequent models until all covariates in the model were significant. The “mgcv” (Wood, 2000) library in R (R Development Core Team, 2010) was used to run the GAM models.

Pollock diet

For pollock that were subsampled from each trawl haul for diet information, lengths and weights were measured and stomach contents were removed. Stomach contents of pollock were identified to the lowest possible taxonomic category (10 prey categories), enumerated, and weighed, and in the case of prey fish, measured. The 10 prey categories were chaetognaths, euphausiids, amphipods, copepods, crab, miscellaneous, and pollock (<60 mm standard length [SL], 60–200 mm SL, ≥ 200 mm SL, and unmeasured). If unmeasured (i.e., too digested to measure), and prey pollock were accompanied by measured prey pollock in the same predator stomach, the unmeasured prey items were assigned the same average length as the measured pollock, otherwise, they were not included in the analyses. The percent frequency of occurrence, partial fullness, and percent body weight of each prey item were calculated for each predator pollock and then averaged over predator size categories (*PredatorLength*) at each station in each year. Predator length categories were 1–19 cm, 20–29 cm, 30–39 cm, 40–49 cm, 50–59 cm, and >60 cm fork length (FL). The presence or absence (PA) of cannibalism (1=cannibalism present; 0=cannibalism absent) was recorded. Percent frequency of occurrence ($\%FO$) was calculated as

$$\%FO = \frac{n_p}{n_f},$$

where n_p = the number of predators that consumed prey type p ; and

n_f = the number of predators with food (f) in their stomachs at that station.

Percent body weight ($\%BW$) was calculated as

$$\%BW = \left(\frac{100}{n_p} \right) \times \sum \left(\frac{W_{pj}}{BW_j} \right),$$

where W_{pj} = the weight of prey type p in predator j ; and
 BW_j = the body weight of predator j .

Partial fullness (PF) was calculated as

$$PF = \%FO \times \%BW.$$

If $\%FO$ was zero, a value of zero was assigned to PF .

Pollock diet models

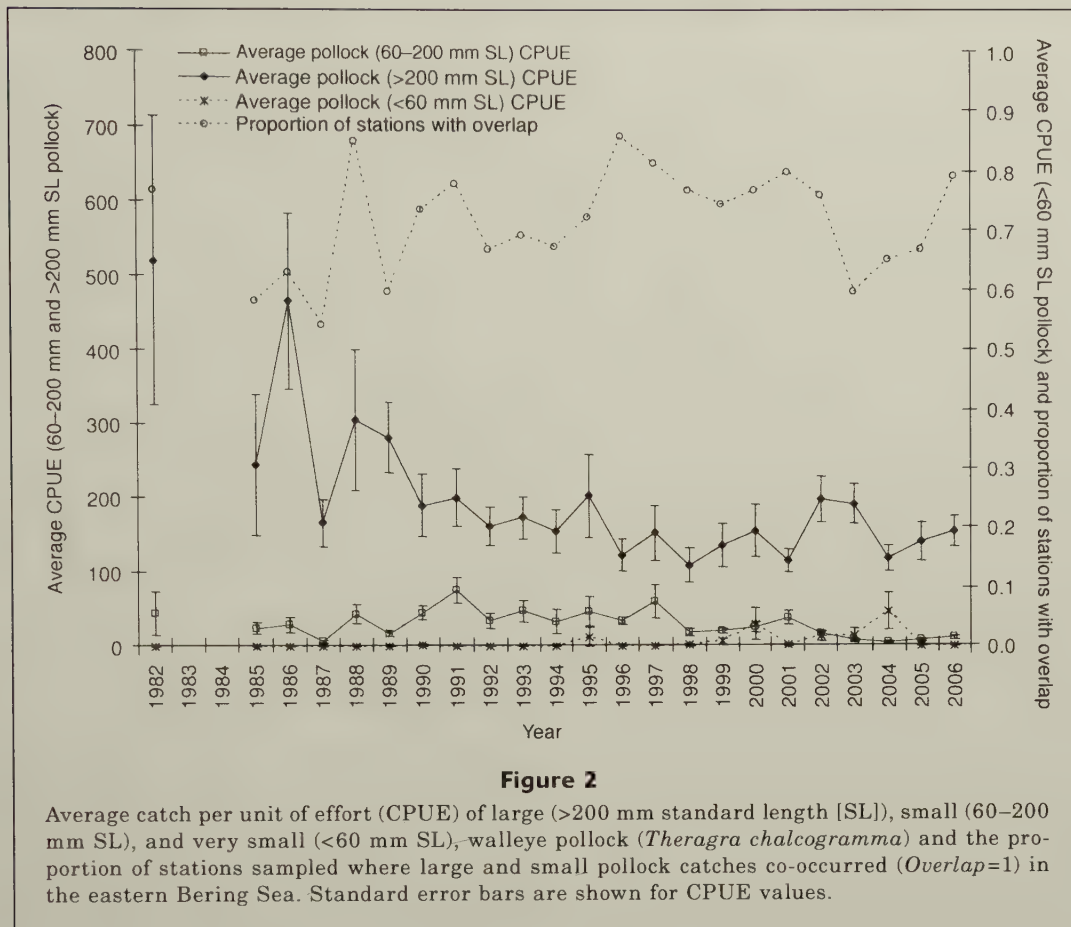
GAMs were also used to explore the relationship between pollock cannibalism on age-1 pollock (60–200 mm SL) and important covariates. There were many zeros in the pollock diet database—a feature typical of this type of data. To address this issue, the first step was to examine whether the presence or absence (PA) of cannibalism (binary response variable) was related to the covariates, therefore, data from all stations were used (stations where cannibalism was and was not found to occur). GAMs were used to explore the relationship between the occurrence of pollock cannibalism on age-1 pollock (PA) and covariates, which were the following: *Year*, *ColdPool*, *PredatorLength*, *Overlap*, *TempDepth* (s[bottom temperature, bottom depth]), and *Location* (s[latitude, longitude]). The time of day that pollock were sampled was not included in GAMs because it likely would not affect evidence of cannibalism as digestion of pollock prey takes longer than 24 hours (Dwyer et al., 1987).

The subsequent step was to determine those factors affecting the relative amount of cannibalism (hereafter referred to as simply the amount of cannibalism) at stations where cannibalism occurred. The amount of cannibalism on age-1 pollock was estimated by weighting $\%FO$, $\%BW$, and PF of prey pollock (60–200 mm SL) by the abundance of large predatory pollock (e.g., $\%FO \times LgPollock$). These weighted estimates were then $\ln+1$ transformed and, hereafter, are referred to as $\%FO_{LgPollock}$, $\%BW_{LgPollock}$, and $PF_{LgPollock}$ and were the response variables assessed. The covariates in these GAMs included *ColdPool*, *PredatorLength*, *TempDepth*, *Location*, and a smoothed *SmallPollock* term (s[*SmallPollock*]). The covariate *Overlap* was not tested as a predictor of the amount of cannibalism because it was assumed that where cannibalism occurred, predator and prey pollock co-occurred. Instead, *SmallPollock* was included as a covariate because the amount of prey pollock available may influence the amount of cannibalism that occurred. The same step-wise model-selection approach that was used for the abundance GAMs was used for the diet GAMs.

Results

Pollock abundance

During 1982–2006, 2754 bottom trawls were conducted on the EBS shelf (Fig. 1). The number of stations sampled ranged from a low of 13 in 1982 to a high of 170 in 1993, with an average of 120 stations sampled per year. Average CPUE of large pollock (>200 mm SL; *LgPollock*) declined until the late 1990s; whereas, average CPUE of small (60–200 mm SL; *SmallPollock*) pollock was high and variable before 1998 and lowest during 2003–2006 (Fig. 2). CPUE of very small pollock (<60 mm SL) was rare (<0.06 ha $^{-1}$) in all years. The proportion of sta-



tions where the *SmallPollock* and *LgPollock* co-occurred (*Overlap*=1) ranged from 0.54 to 0.86, and there was no temporal trend; minimum values occurred in 1985, 1987, 1989, and 2003 and peak values occurred in 1988 and 1996 (Fig. 2).

Pollock abundance models

Of the four covariates examined (*TempDepth*, *Location*, *Year*, and *ColdPool*), *TempDepth* and *Location* were both significant predictors of large pollock CPUE (*LgPollock*), small pollock CPUE (*SmallPollock*), and *Overlap*. In addition, *Year* and *ColdPool* were significant predictors of *LgPollock*, *Year* was a significant predictor of *SmallPollock*, but neither *Year* nor *ColdPool* were significant predictors of *Overlap*. Significant covariates explained 51.0%, 22.2%, and 17.1% of the deviance in models for *LgPollock*, *SmallPollock*, and *Overlap*, respectively (Table 1). Over the years examined, *LgPollock* and *SmallPollock* decreased significantly (Table 1, Fig. 2). *LgPollock* was highest in the northwest, outer, and middle domains of the EBS and was lowest in the east EBS and in the coastal domain (Fig. 3). Large pollock tended to avoid the cold pool and *LgPollock* was highest in bottom temperatures greater than 1°C with bottom depths of 75–150 m (Fig. 3). Small pollock were found primarily

in the middle and outer domains of the northwest EBS, but had a broader distribution than large pollock (Fig. 3). Small pollock (*SmallPollock*) were found at slightly shallower bottom depths (50–125 m) than large pollock, but there was considerable overlap in *SmallPollock* and *LgPollock* bottom depth ranges (Fig. 3). Small pollock were found in a wider range of bottom temperatures than the range found for large pollock. *SmallPollock* was high at temperatures between –1°C to 5°C (Fig. 3). *Overlap* was, as expected from *LgPollock* and *SmallPollock* trends, highest in the middle and outer domains of the northwest EBS, at bottom depths less than 100 m, with bottom temperatures between 0°C and 5°C (Fig. 3).

Pollock diet

Pollock sampled in the EBS during 1982–2006 consumed primarily euphausiids and copepods (Fig. 4). Other important prey included amphipods, chaetognaths, crabs, and pollock (Fig. 4). Miscellaneous prey items included other fish, larvaceans, mysids, shrimp, and other prey (Fig. 4). The %BW of prey pollock (60–200 mm SL) consumed decreased during the time period examined (Fig. 4). Prey pollock 60–200 mm SL were the dominant-size pollock consumed by predator pollock measuring >19 cm FL (Fig. 5). Prey pollock >200 mm

Table 1

Final general additive models from a backward selection process and the associated adjusted r^2 values, deviance explained (%), and sample size (n). b =intercept, s =nonparametric smoothing function, E =error term. Response variables are: $LgPollock$ = $\ln+1$ transformed predator walleye pollock (*Theragra chalcogramma*) (>200 mm standard length [SL]) catch per unit of effort (CPUE) in the NMFS bottom trawl survey; $SmallPollock$ = $\ln+1$ transformed prey pollock (60–200 mm SL) CPUE in the bottom trawl survey; PA =binomial presence or absence of cannibalism; $\%FO_{LgPollock}$, $\%BW_{LgPollock}$, and $PF_{LgPollock}$ = $\ln+1$ transformed percent frequency of occurrence, percent body weight, and partial fullness of prey pollock in predator pollock stomachs, weighted by $LgPollock$, respectively. Covariates were the following: *Location*, a smoothed station location covariate, $s(\text{latitude, longitude})$; *TempDepth*, a smoothed bottom temperature and depth covariate, $s(\text{bottemp, botdepth})$; *ColdPool*, a binomial covariate indicating presence or absence of the cold pool; *PredatorLength*, a predator length categorical covariate; *Overlap*=binomial covariate indicating the spatial overlap between predator and prey pollock; $s(SmallPollock)$, a smoothed *SmallPollock* covariate.

Model type	Final model formulation	r^2 (adjusted)	Deviance explained (%)	n
CPUE	$LgPollock = b + s(Location) + s(TempDepth) + Year + ColdPool + E$	0.50	51.0	2754
CPUE	$SmallPollock = b + s(Location) + s(TempDepth) + Year + E$	0.21	22.2	2754
CPUE	$Overlap = b + s(Location) + s(TempDepth) + E$	0.19	17.1	2754
Diet	$PA = b + s(Location) + s(TempDepth) + Year + PredatorLength + Overlap + E$	0.13	21.1	7479
Diet	$\%FO_{LgPollock} = b + s(Location) + s(TempDepth) + s(SmallPollock) + E$	0.52	55.1	610
Diet	$\%BW_{LgPollock} = b + s(Location) + s(TempDepth) + s(SmallPollock) + PredatorLength + E$	0.63	65.5	610
Diet	$PF_{LgPollock} = b + s(Location) + s(TempDepth) + s(SmallPollock) + PredatorLength + E$	0.55	58.3	610

SL were generally consumed by predators measuring >60 cm FL and prey pollock <60 mm SL were consumed in small proportions by all sizes of predator pollock (Fig. 5). Additional pollock were consumed but could not be included in the analyses because they were too digested to obtain length measurements (Fig. 5). The occurrence of cannibalism in the EBS varied spatially and temporally (Figs. 4 and 6). The proportion of stations where cannibalism occurred ranged from 0.02 (in 2003 and 2004) to 0.18 (in 1990) and decreased during the years examined (Figs. 4 and 6). Fewer samples were collected in 1982, 1985, and 1986 (Fig. 6).

Pollock diet models

The presence or absence of pollock cannibalism on age-1 pollock (60–200 mm SL) was significantly affected by most covariates examined, *TempDepth*, *Location*, *Year*, *PredatorLength*, and *Overlap* (excluding *ColdPool*), and the model explained 21.1% of the deviance (Table 1). The occurrence of cannibalism was highest in the northwest EBS and covered portions of all depth domains, but primarily in the middle and outer domains (Fig. 7). The presence of cannibalism generally occurred most often at bottom temperatures between 0°C and 5°C (Fig. 7).

Where cannibalism occurred, the amount of cannibalism ($\%FO_{LgPollock}$, $\%BW_{LgPollock}$, and $PF_{LgPollock}$) on age-1 pollock (60–200 mm SL) was significantly affected by *TempDepth*, *Location*, *SmallPollock*, and *PredatorLength* (except *PredatorLength* in the $\%FO_{LgPollock}$ GAM). Covariates that were not significant in the diet GAMs were *ColdPool* (except in the $\%FO_{LgPollock}$ GAM) and *Year*. Significant covariates explained 55.1%, 65.5%, and 58.3% of the deviance in models for $\%FO_{LgPollock}$, $\%BW_{LgPollock}$, and $PF_{LgPollock}$, respectively

(Table 1). Generally, the amount of cannibalism was highest in the northwest EBS, in the outer half of the middle depth domain and in the outer domain (Fig. 7). The amount of cannibalism increased with increasing temperatures between 1°C and 5°C and with increasing bottom depths (Fig. 7). In addition, the amount of cannibalism increased with increasing abundance of small pollock (*SmallPollock*) and increasing predator size (*PredatorLength*; Fig. 7).

Discussion

The GAM approach, used in this study, enabled us to test hypotheses regarding pollock distribution and cannibalism. Results from this study showed that the distributions of large cannibalistic pollock and vulnerable age-1 prey pollock were affected by environmental factors and varied among years. Large pollock were generally found in deeper waters of the outer and middle domains and they avoided the cold pool; whereas, age-1 pollock were more broadly distributed and were generally found in slightly shallower (but overlapping) depths in the northwest middle and outer domains and in cooler temperatures. These results are consistent with what has been found in previous studies regarding pollock distribution in the EBS (Swartzman et al., 1994; Kotwicki et al., 2005; Mueter et al., 2011). The area where cannibalism on age-1 pollock could potentially have occurred (area of overlap between predator and prey pollock) was in the middle and outer domains of the northwest EBS at depths less than 100 m and at bottom temperatures between 0°C and 5°C.

Although diet samples represent snapshots in time, the spatial coverage over the 23 years examined pro-

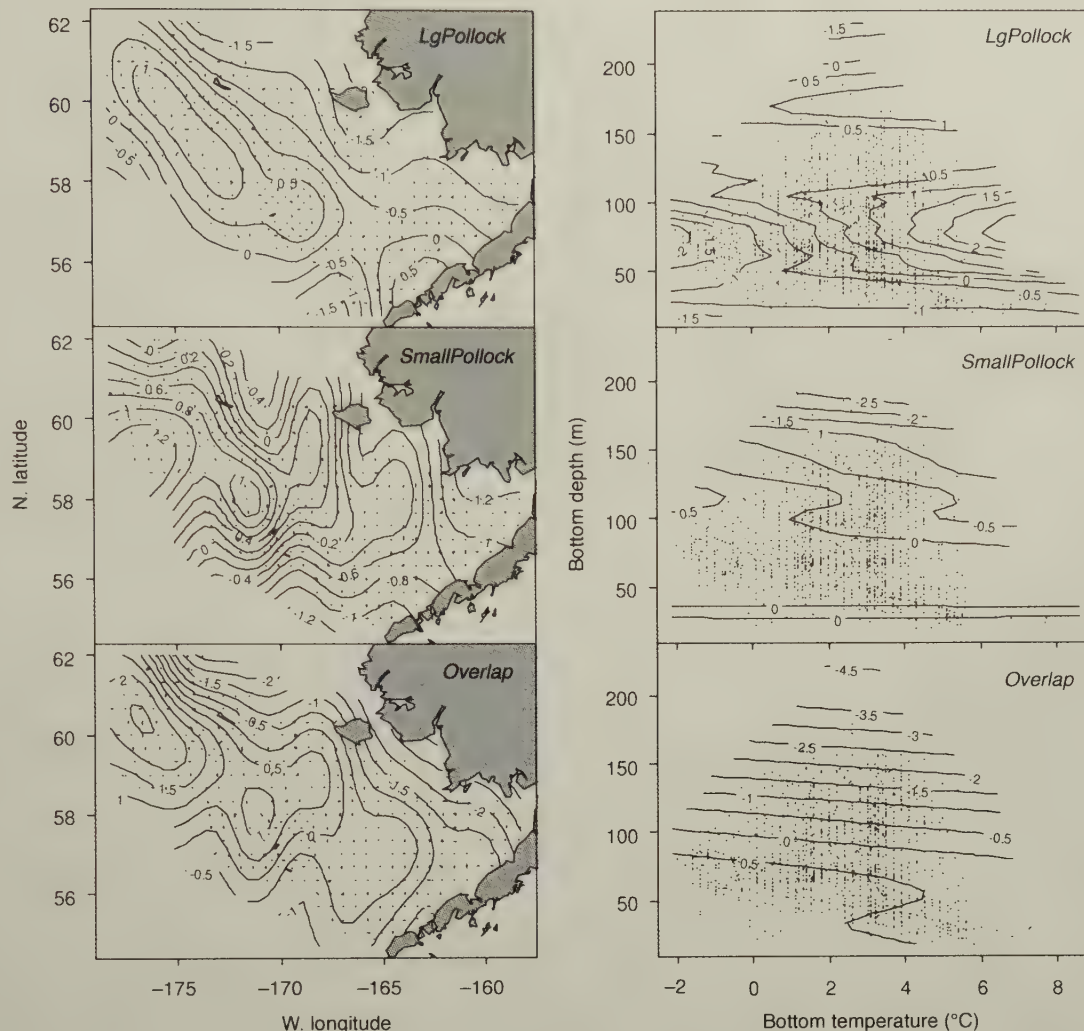


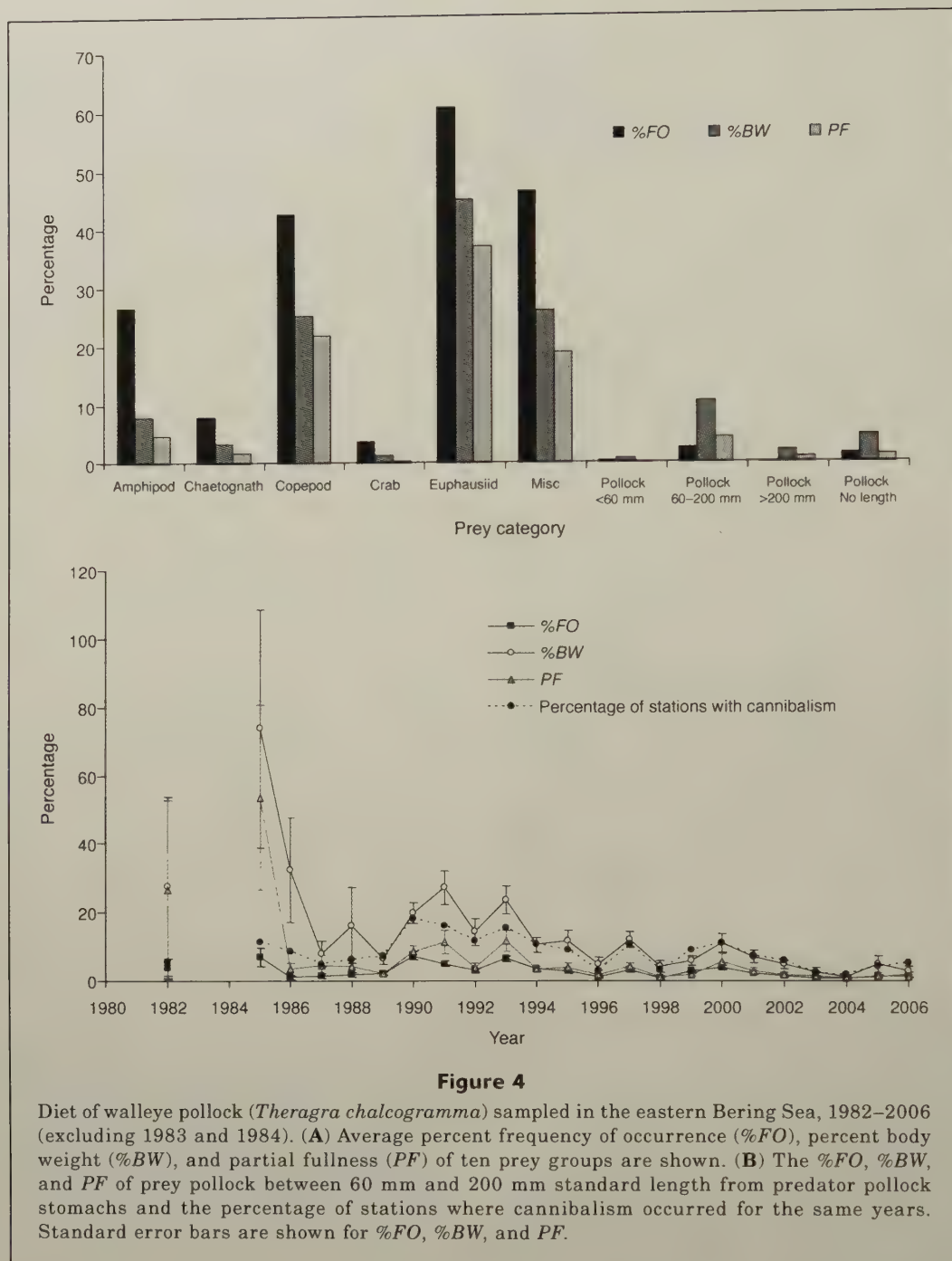
Figure 3

General additive model results showing the additive effect of two smoothed covariates on walleye pollock (*Theragra chalcogramma*) catch per unit of effort (CPUE) from the National Marine Fisheries Service bottom trawl survey on the eastern Bering Sea shelf, during 1982–2006 (excluding 1983 and 1984). The two smoothed covariates are location (North latitude and West longitude; left column) and habitat (bottom depth [m] and bottom temperature [°C]; right column). Shown are the additive effect of these two smoothed covariates on $\ln+1$ transformed large (>200 mm standard length [SL]) pollock CPUE (LgPollock; top row), $\ln+1$ transformed small (60–200 mm SL) pollock CPUE (SmallPollock; middle row), and the co-occurrence of the two (Overlap; bottom row).

vided an opportunity to improve our understanding of factors affecting cannibalism. As found in previous studies (Dwyer et al., 1987; Lang and Livingston, 1996), the diet of pollock comprised mainly euphausiids and copepods, and cannibalism was prevalent in many samples. The occurrence and amount of cannibalism in the EBS decreased during 1982–2006 and were affected by environmental factors. As expected, the area of overlap between predator and prey pollock was the area where the occurrence of cannibalism was most frequently observed, on the northwest middle and outer domains. The greatest amount of cannibalism occurred in the

offshore portion of the middle domain and the outer domain and increased with temperatures (between 1°C and 5°C) and bottom depths. Moreover, the amount of cannibalism increased with higher prey pollock abundances and increasing predator sizes.

We found that, in the summer, the occurrence of cannibalism was related to location with its associated bottom depth and temperature, the presence of the cold pool, and the overlap between adult and age-1 pollock. The less frequent occurrence of cannibalism in the cold pool reflected the tendency of large pollock to avoid these areas. Where cannibalism did occur, the



amount that occurred was also related to location and its associated bottom depth and temperature, and the abundance of age-1 prey pollock, but was not related to the presence of the cold pool. Mueter et al. (2006) also found no evidence that the presence of the cold pool was related to total predation mortality (by multiple species of predators) on age-1 pollock, as estimated by a multispecies virtual population analysis; instead total predation mortality of age-1 pollock was related to the abundance of adult pollock and the spatial association between juveniles and adults.

The spatial overlap variable between predator and prey pollock in this study was a measure of horizontal overlap and did not account for potential differences in vertical distribution. Age-0 pollock are found above the thermocline in the summer, and several studies have examined the hypothesis that water column stratification separates them from cannibalistic adults during the summer (Bailey, 1989; Swartzman et al., 1994). In the summer, the majority of prey pollock available to large fish are age-1 pollock, which are often found near the bottom (Duffy-Anderson et al., 2003). The spatial

overlap variable in this study indicated stations where age-1 pollock and adult pollock were caught in the same bottom trawl haul, and hence, both age groups of pollock would have occupied similar depths at these stations. Spatial overlap between age 0 and other pollock age classes could not be estimated in this study because the bottom trawl net does not catch small individuals efficiently and age-0 pollock are distributed higher in the water column. It is worth noting, that we also tested the effect of an indicator of water column stratification (and its interaction with depth) on the occurrence and amount of cannibalism. The water column stratification indicator was calculated as residuals from a linear regression between the day of year and the temperature difference between surface and bottom waters (an indicator of the level of water column stratification). The results were similar to those of the GAM models presented here that included bottom temperature instead of water column stratification, and in fact the stratification variable was significantly correlated with bottom temperature. The most parsimonious model, therefore, was based on bottom temperature and is the only result presented in this study.

A confounding factor in our models was the day of year that samples were collected. The dates of the NMFS EBS shelf bottom trawl surveys varied annually and the sampling started in the southeast EBS and generally proceeded northwestward. Start dates for the surveys usually occurred in the first week of June of each year, but ranged from 24 May in 1999 to 19 June in 1986. End dates for the survey were typically in the last week of July in each year, but ranged from 11 July in 1982 to 14 August in 1985. Day of year is, therefore, confounded with the factors year and location to some degree in our models, and as such, the effect of sample day cannot be completely separated from these factors.

Water temperatures and water column stratification can affect multiple biological processes on the EBS shelf and have implications for the zooplankton community and predation on pollock. In warm years, stratification of the water column tends to occur earlier and result in stronger summer stratification (Coyle et al., 2008; McKinnell and Dagg, 2010). During warm years, both the zooplankton community and diet of age-0 pollock tend to be dominated by smaller copepods, whereas, in cold years, diets are dominated by larger copepods and euphausiids (Baier and Napp, 2003; Coyle et al., 2008, 2011; Hunt et al., 2011). Accompanying the reduction in prey availability in warm years, more cannibalism on age-0 pollock was observed in warm than in cold years (Moss et al., 2009; Coyle et al., 2011). Warm years also bring more potential pollock predators, such as arrowtooth flounder (*Atheresthes stomias*), northward and onto the EBS

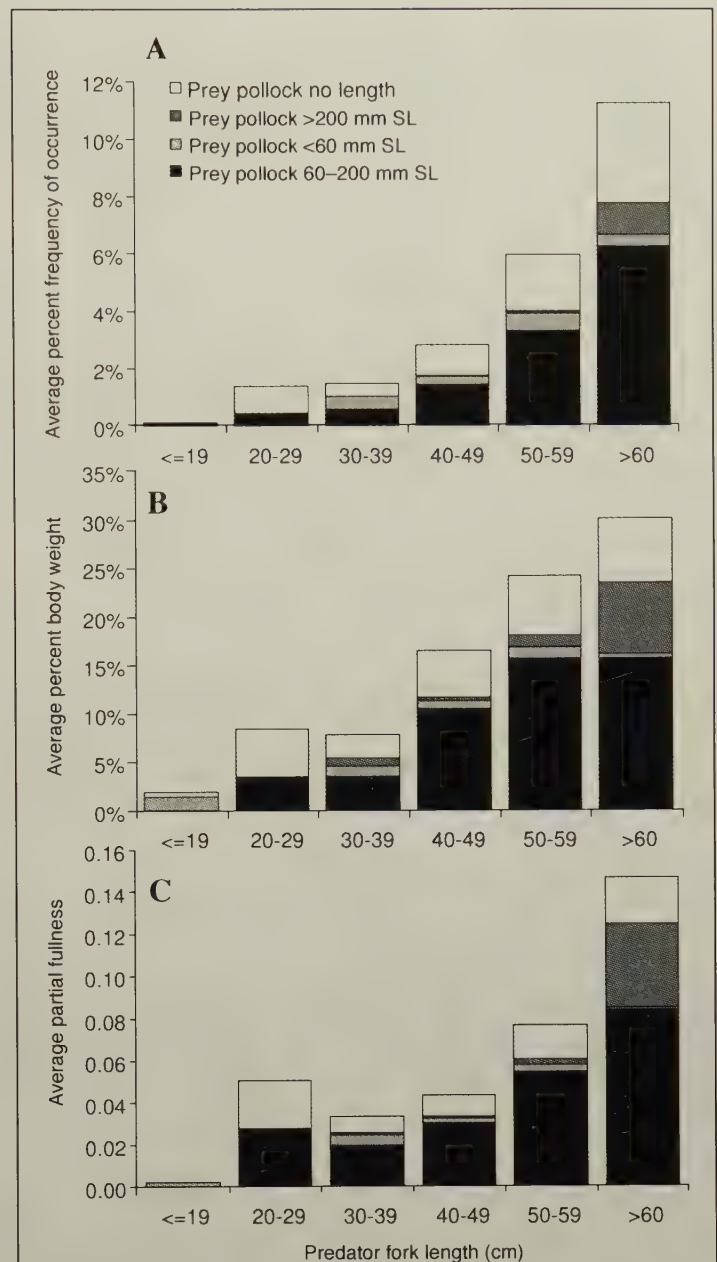


Figure 5

Walleye pollock (*Theragra chalcogramma*) cannibalism by predator pollock fork length (cm) in the eastern Bering Sea during 1982–2006 (excluding 1983 and 1984). (A) Average percent frequency of occurrence (%FO), (B) average percent body weight (%BW), and (C) average partial fullness (PF) of prey pollock measuring less than 60 mm, 60–200 mm, and greater than 200 mm standard length, and those prey pollock that were too digested to measure (no length).

shelf (Mueter and Litzow, 2008; Ianelli, et al., 2011). We found that the cold pool affected the distribution of large pollock and, therefore, in years with a large cold pool, a reduced overlap in distribution of large and small pollock may reduce the occurrence of can-

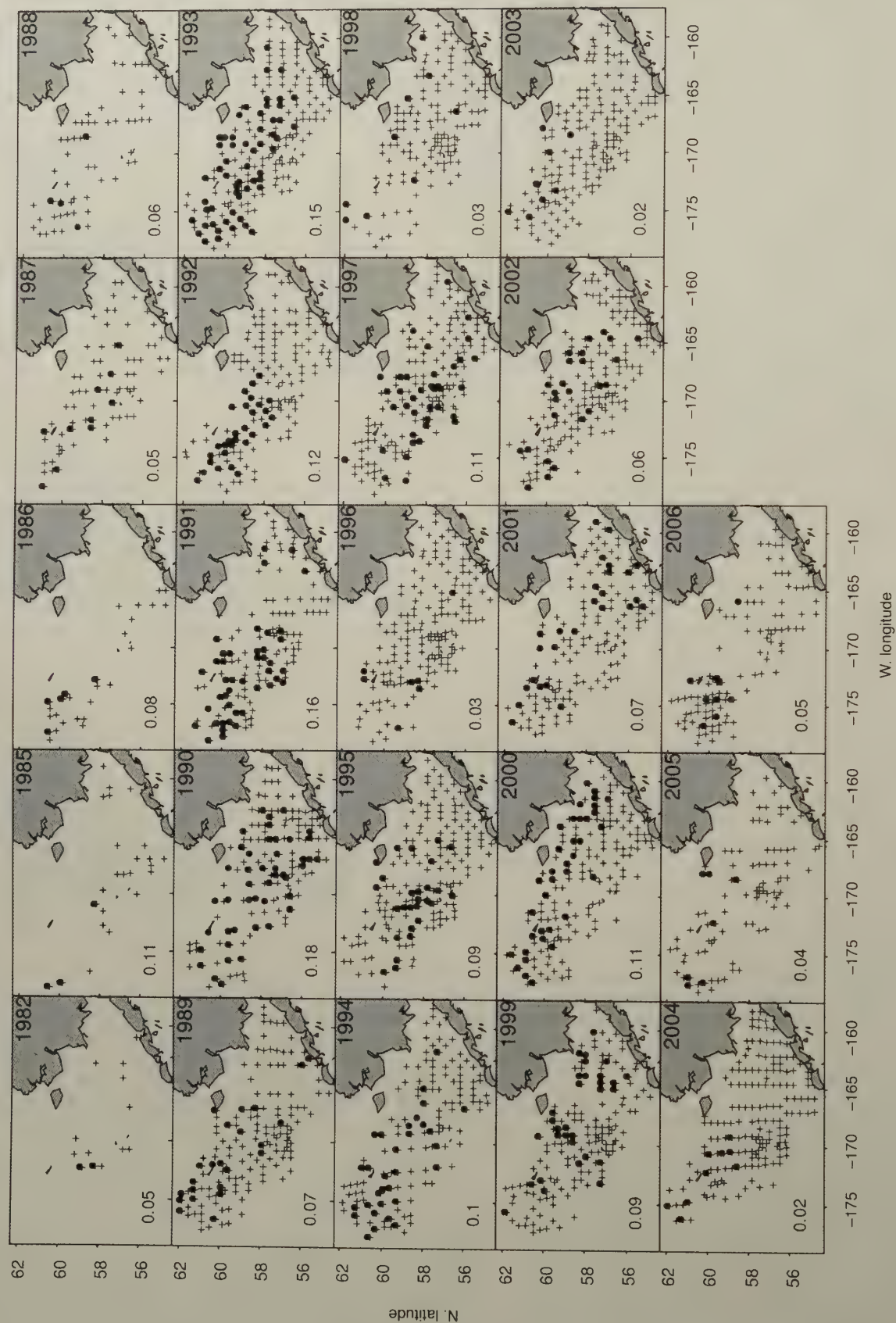


Figure 6

Locations where walleye pollock (*Theragra chalcogramma*) cannibalism was observed (circles) in the eastern Bering Sea during 1982–2006 (excluding 1983 and 1984). A “+” indicates stations where samples were collected and examined, but no cannibalism was observed. Numbers in lower left corners of maps indicate the proportion of stations with cannibalism each year.

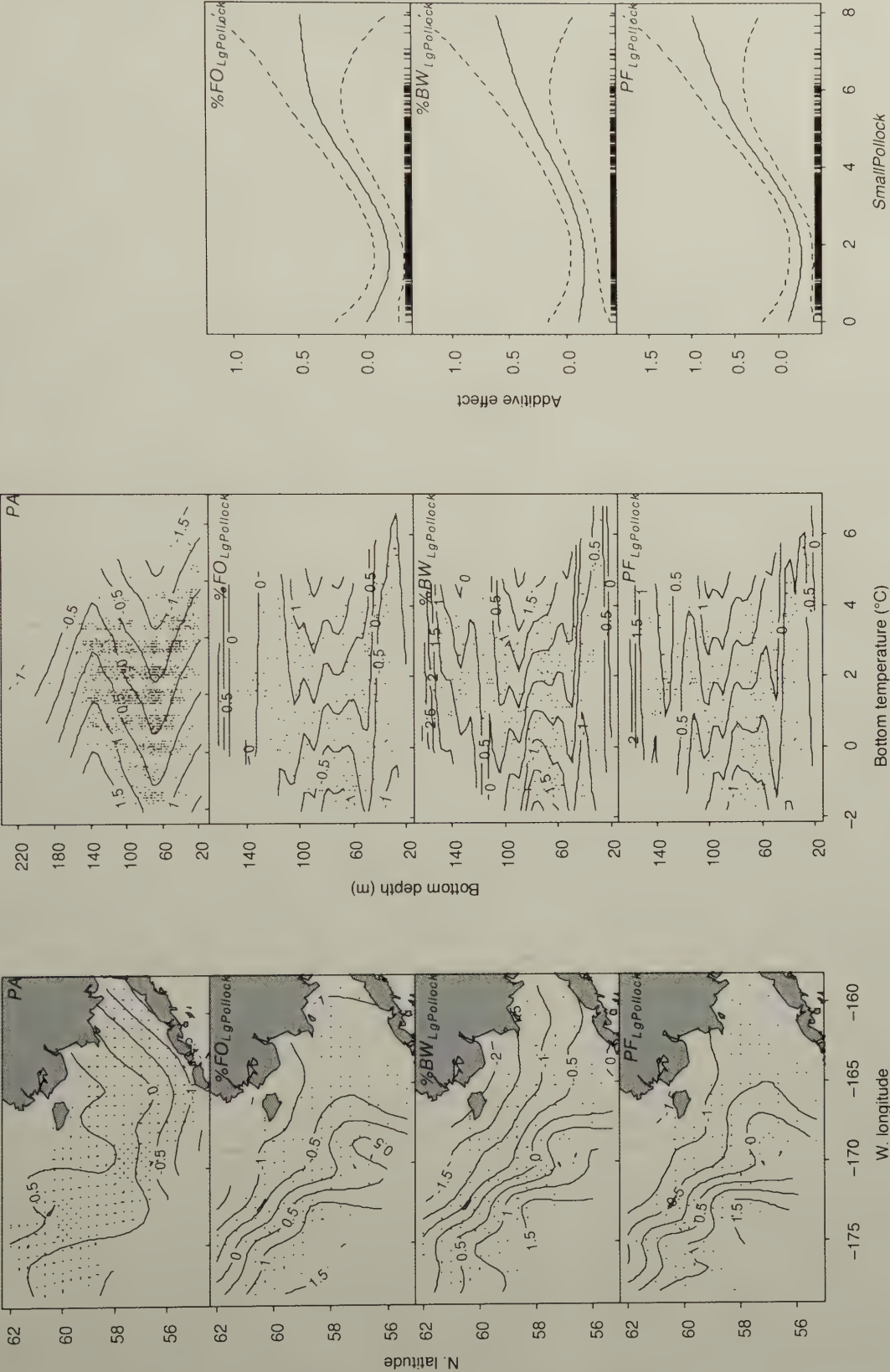


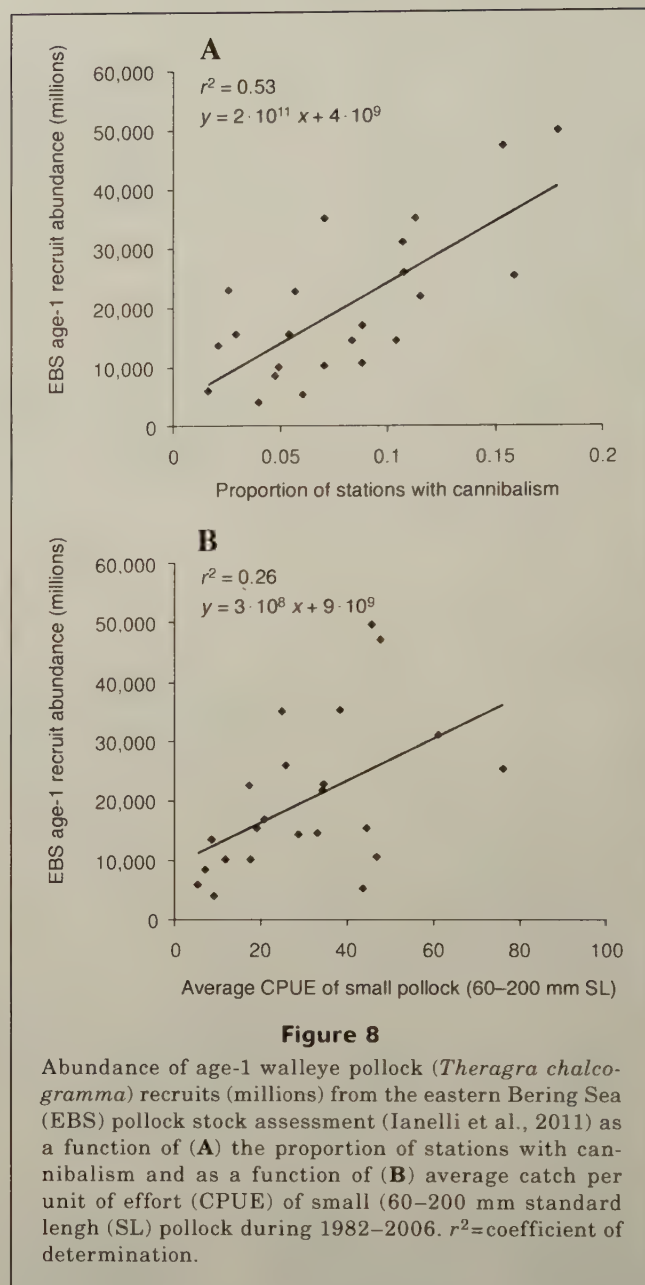
Figure 7

General additive model results showing the additive effect of 3 smoothed covariates on walleye pollock (*Theragra chalcogramma*) cannibalism from the National Marine Fisheries Service bottom trawl survey on the eastern Bering Sea shelf, during 1982–2006 (excluding 1983 and 1984). The 3 smoothed covariates are location (North latitude and West longitude; left column), habitat (bottom depth [m] and bottom temperature [°C]; middle column), and ln+1 transformed small (60–200 mm standard length [SL]) pollock catch per unit of effort (CPUE) (*SmallPollock*; right column). Shown, are the additive effect of these 3 smoothed covariates on the presence or absence (*PA*) of cannibalism (top row), percent frequency of occurrence of small prey pollock in predator pollock stomachs, weighted by the CPUE of large (>200 mm SL) pollock (*%FO_{LgPollock}*; second row), percent body weight of small prey pollock in predator pollock stomachs, weighted by the CPUE of large pollock (*%BW_{LgPollock}*; third row); and the partial fullness of small prey pollock in predator pollock stomachs, weighted by the CPUE of large pollock (*%PF_{LgPollock}*; bottom row).

nibalism. Also, the amount of cannibalism generally increased with increasing temperatures; however, the amount of cannibalism on age-1 pollock was low during the warm years of 2000–2005, possibly because of the reduced abundance of both large and small pollock. The revised oscillating control hypothesis (Hunt et al., 2002, 2011) and studies of age-0 pollock diet (Moss et al., 2009; Coyle et al., 2011) predict that during warm years there is increased top-down control of age-0 pollock (<150 mm) in the upper water column, but these studies do not address cannibalism of demersal, age-1 pollock.

In the fall, cannibalism on age-0 pollock is high and influences year-class strength (Dwyer et al., 1987) and age-1 pollock cannibalism occurs, but is less prevalent (Dwyer et al., 1987; Bailey, 1989). During summer months, age-1 pollock comprise the majority of cannibalized prey pollock (Dwyer et al., 1987). Mueter et al. (2006) found that predation (including cannibalism) of age-1 pollock by multiple predator species explained up to 76% of the variability in survival estimates (log-transformed recruit per spawner biomass) and concluded that cannibalism was a significant factor affecting the survival of pollock. To explore the possibility that cannibalism on age-1 pollock in summer months affects year-class strength, we examined the relationship between summer cannibalism and the abundance of age-1 recruits from the stock assessment model (Ianelli et al., 2011). In our study, there were significant, positive relationships between the number of age-1 recruits and 1) the amount of pollock cannibalized ($r^2=0.40$, 0.49, and 0.39; and $P=0.001$, <0.001, and 0.001, for $\%BW_{LgPollock}$, $\%FO_{LgPollock}$, and $PF_{LgPollock}$, respectively) and 2) the proportion of stations where cannibalism was present ($r^2=0.53$, $P<0.001$; Fig. 8). The occurrence and amount of cannibalism increased significantly with increasing age-1 recruit abundances, suggesting that cannibalism on age-1 pollock is not controlling recruitment, and strong year classes may overwhelm the capacity for cannibalism. These results imply that summer cannibalism of age-1 pollock, a top-down process, is not the sole determinant of age-1 pollock abundance.

In this study, we found that the occurrence and amount of cannibalism observed in the survey diet samples were good indicators of year-class strength, because when there were more age-1 pollock consumed, the predicted abundance of age-1 recruits in the stock assessment model was higher. The relationship between the small pollock CPUE (*SmallPollock*) in the bottom trawl survey and the abundance of age-1 recruits from the stock assessment model is also significant and positive ($r^2=0.26$, $P=0.013$; Fig. 8); however, *SmallPollock* does not explain as much variability in age-1 recruit abundance from the stock assessment model as the occurrence of cannibalism does. This result is likely due to the selectivity of the survey bottom trawl net, from which smaller fish may escape. Adult pollock are thus better samplers of age-1 pollock because the occurrence and amount of cannibalism is indicative of age-1 abundance.



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Abstract—We studied a small artisanal fishery for the spotted eagle ray (*Aetobatus narinari*) off Margarita Island in northeastern Venezuela. We analyzed data from 413 fishing trips directed at *A. narinari* over a 29-month sampling period (August 2005–December 2007). These trips yielded 55.9 metric tons and 1352 individuals from which a subsample of 846 females and 321 males was used for biological data. Maximum fishing effort and landings occurred between February and May, and catch per unit of effort was highest between December and February and between July and October with an overall average of 3 individuals and 133 kg per trip. The overall sex ratio was significantly different from 1:1 with a predominance of females. Females ranged in size with disc widths (DW) from 64 to 226 cm. Males ranged in size between 97 and 190 cm DW. There was no statistically significant difference between male and female length-weight relationships. Mean fecundity was estimated at 3.09 embryos per female, and the largest embryo measured 44.5 cm DW. Females in different maturity stages were found in all months, except November 2007, the month when all females were immature. Postgravid females occurred mainly during the periods of January–May and July–October. Mean length (L_{50}) at maturity was estimated at 129.2 cm DW for males and 134.9 cm DW for females. This study provides much needed information on the biology and life history of *A. narinari* for the management of an intensive, directed, small-scale fishery for this little known species in northeastern Venezuela.

Exploitation and reproduction of the spotted eagle ray (*Aetobatus narinari*) in the Los Frailes Archipelago, Venezuela

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The spotted eagle ray (*Aetobatus narinari*) until recently had been considered a cosmopolitan species distributed in tropical and warm, temperate waters. However, recent genetic analysis indicates that what was thought to be a single species is actually a species complex with at least 2 separate species, 1 for the western and central Pacific and at least 1 for the eastern Pacific and central Atlantic (Richards et al., 2009; Schluessel et al., 2010a). Furthermore, White et al. (2010) in a taxonomic review of this species complex established that *Aetobatus ocellatus* is the valid species name for the Indo-West Pacific region. Both species are very similar morphologically, and the major physical difference occurs in the background dorsal coloration (White et al. 2010). Because most previous studies have been conducted in the Indo-Pacific region, this taxonomic redefinition significantly reduces the available scientific literature on the biology and fisheries of *A. narinari*. In the western Atlantic, *A. narinari* is found from Chesapeake Bay and Bermuda to the Gulf of Mexico, Caribbean Sea, and south to Brazil (Cervigón and Alcalá, 1999).

Aetobatus narinari is observed usually in coastal environments, such as bays and coral reefs, and occasionally in estuarine habitats. In inshore waters, it has been observed to depths of 60 m and is known to travel long distances across open waters (International Union for the Conservation of Nature [IUCN] Red List of Threatened Species, vers. 2011.2 [Available from <http://www.iucnredlist.org/apps/redlist/details/39415/0>, accessed July 2011]).

The spotted eagle ray is a benthic feeder with a diet consisting mainly of bivalve and gastropod molluscs (Randall, 1967, for *A. narinari*; Schluessel et al., 2010b for *A. ocellatus*), but it also consumes cephalopods, crustaceans, and teleost fishes. It is found to be solitary or to swim in large schools of up to several hundred individuals (McEachran and de Carvalho, 2002).

Little is known about the biology and reproduction of *A. narinari*. Individuals reach large sizes, and if the distance between both extremes of the pectoral fins or disc width (DW) is used as a measure of length, the maximum reported length is 230 cm DW; most reported individuals, how-

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ever, are usually less than 140 cm DW (McEachran and de Carvalho, 2002). *Aetobatus narinari* has a matrotrophic mode of embryonic gestation and fetal nutrition occurs by lipid histotrophy; and this species has low fecundity with 1–4 pups per litter (McEachran and de Carvalho, 2002). Length at maturity has been estimated for *A. ocellatus* in the western Pacific and Indian Oceans, but to our knowledge there is only one previous report for *A. narinari* in the Atlantic Ocean (Dubick, 2000).

The spotted eagle ray is categorized as near threatened throughout its range by the IUCN and studies on its catch, abundance, and reproductive biology have been recommended. However, the conservation status of this species complex needs to be reviewed in light of the recent taxonomic changes (White et al., 2010).

The spotted eagle ray is considered of minor commercial importance, but it is caught as bycatch with different fishing-gear types, such as trawls, trammel nets, and longlines (Trent et al., 1997; Stobutzki et al., 2002; Grijalba-Bendeck et al., 2007). In the southeastern Gulf of Mexico, Cuevas-Zimbrón et al. (2011) reported a small-scale, directed fishery for *A. narinari*.

In Venezuela, *A. narinari* is considered a common species, particularly in insular areas with coral reef cover, such as the Los Roques Archipelago in the north central part of this country (Cervigón, 2005) and the islands of Coche, Cubagua, and Margarita off the northeastern coast. *Aetobatus narinari* is readily marketable fresh or salted and is particularly appreciated in eastern Venezuela where it is the main part of a typical dish (Cervigón and Alcalá, 1999). At Margarita Island, we studied a small, directed fishery to generate information that may contribute to the management and conservation of this little known and potentially vulnerable species.

Materials and methods

Study area and fishing fleet

The insular state of Nueva Esparta in northeastern Venezuela is formed by the islands of Coche, Cubagua, and Margarita. Margarita and Coche each have an important concentration of small-scale fishing communities. Puerto Fermín, also known as El Tirano, is a traditional fishing community in northeastern Margarita. Fisheries at Puerto Fermín mainly use trap gear, but a small number of boats target *A. narinari* around the nearby Los Frailes Archipelago (63°46′–63°43′N latitude, 11°14′–11°12′W longitude). This specialized fleet is composed of 6 wooden boats with outboard engines, known locally as *peñeros*, with 2 boats that fish only occasionally and the other 4 boats that fish every month. This fleet uses bottom-set gillnets that vary between 200 and 500 m in length, between 8 and 10 m in height, and between 30 and 39 cm in mesh size. Nets are set in the evening between 1600 and 1830 hours and lifted the following day between 0500 and 0900 hours. Detailed descriptions of boats

and fishing gear for this area may be found in Méndez-Arocha (1963), Ginés et al. (1972), Iriarte (1997), and González et al. (2006).

Field sampling and biological data

Fishing for the spotted eagle ray is carried out every month during a period of ~10 days around the full moon. Samples were taken during such periods over the 29 months between August 2005 and December 2007. We registered the number of boats fishing daily, the number of individuals landed, and their weight in kilograms. The retained indices of relative abundance were the number of individuals captured per trip and total kilograms collected per trip.

To determine the size of *A. narinari*, DW was measured as a proxy for total length in ray (total length cannot be reliably measured because tails can be damaged during capture) (Last and Stevens, 2009); therefore, unless otherwise specified, all subsequent references to *length* in this article refer to DW. Specimens were weighed with a field balance with a weighing capacity of 200 kg and a precision of 1 kg. Sexes were differentiated by the presence of claspers in males and their absence in females.

Male sexual maturity was verified by checking claspers visually and by touch. Individuals were considered mature when claspers were strongly calcified, rotated easily around the base, had an extensible distal extreme (Pratt, 1979; Conrath, 2005), and showed the presence of seminal fluid (Bizarro et al., 2007).

Female maturity was identified by macroscopic observation of reproductive organs in fish at the landing site in Puerto Fermín. Individuals were considered mature (nongravid) or immature based on the presence or absence of fully developed ova in the ovaries (Conrath, 2005). When possible, embryos from gravid females were counted, sex was determined, and DW measured (in centimeters). Postgravid females were recognized by the presence of a well-developed, large, highly vascularized uterus (Conrath, 2005) that, in most cases, contained intrauterine milk or histotrophe. Some of these postgravid females were considered gravid if fishermen indicated that abortions occurred during hauling of the specimens.

Other measurements, such as clasper length, follicle diameter, uterus width, gonad weight, and liver weight could not be obtained because of the speed at which animals were cut and sold on arrival at the landing site in Puerto Fermín. The swift cutting and selling of rays also prevented biological measurements of some of the landed individuals and embryos.

Data analysis

Population structure was analyzed by sex from length-frequency data. Length-weight relationships, of the form $Wt = aDW^b \exp^e$, were established for males and females separately, where a is the intercept, b is the slope, $\exp^e$ represents the residual error, and Wt stands

for weight. Statistical differences between these functions were established with an *F*-test from the analysis of the residual sum of squares, in a manner similar to the one proposed by Chen et al. (1992).

Differences in the proportions by sex for juveniles, adults, and embryos were estimated monthly by using a chi-square test (χ^2) with Yates's correction (Zar, 1996). To determine the reproductive period for *A. narinari*, the proportions of immature and mature males and the proportions of females in different reproductive stages were examined monthly. Also, we estimated the mean and range of embryo lengths. Fecundity was determined by the number of embryos per uterus, and the size at birth was determined from the largest observed embryos.

The length at which 50% of individuals were mature was estimated for males and females by a logistic function:

$$M_f = \frac{1}{1 + \exp^{-a(L_i - b)}}$$

where M_f = the fraction of mature individuals;

a = the change in slope of M_f as a function of length intervals (L_i); and

b = the length at 50% maturity (L_{50}).

Parameter estimates for a and b were obtained by maximizing the log-likelihood function and assuming a binomial error distribution (Welch and Foucher, 1988). Confidence intervals were estimated by joint likelihood profiles (Venzon and Moolgavkar, 1988). Parameter estimates and confidence intervals were calculated with the statistical software R, vers. 2.13.0 (R Development Core Team, 2011¹).

Results

Effort, catch, and catch per unit of effort

In the 29-month sampling period, 413 fishing trips directed at *A. narinari* were analyzed. Sampling yielded 55.9 metric tons and 1352 individuals. A subsample of 1167 individuals, 846 females and 321 males, was measured for biological data.

The largest numbers of fishing trips per month, with more than 30 trips per month, were observed during the months of February, April, and May 2006 and February 2007 (Fig. 1A). For 2006 and 2007, the number of trips in the first 6 months of each year corresponded to 73% and 69% of the total number of trips made in each year, respectively. Highest catches in 2006 and 2007 occurred in months with most fishing effort (Fig. 1B). Catches in the first 6 months represented 81% and 62% of total annual catch for the years 2006 and 2007, respectively. Approximately

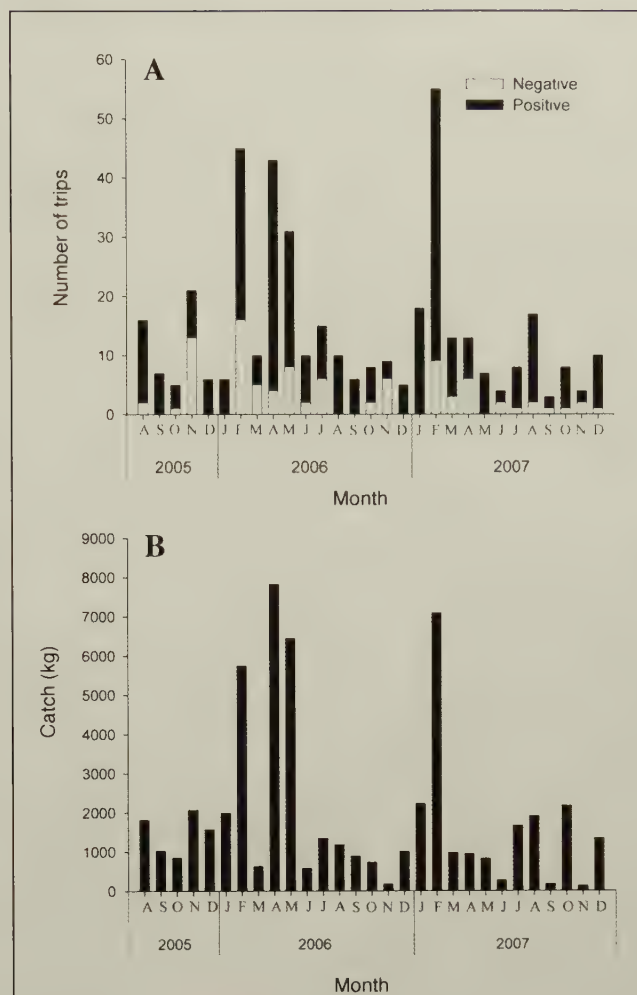


Figure 1

(A) Fishing effort in number of trips per month directed at spotted eagle ray (*Aetobatus narinari*) in the Los Frailes Archipelago in northeastern Venezuela during the period from August 2005 to December 2007. Negative=trips with no catch of *A. narinari*; Positive=trips with catch of *A. narinari*. (B) Catch of *A. narinari* by weight (kg) per month in the fishery in the Los Frailes Archipelago during the period from August 2005 to December 2007.

23% of the total number of trips during the study period resulted in no catch. In all years of this study, November was the month with the highest proportion (>50%) of trips with no catch.

Data on catch per unit of effort (CPUE) in number of fish caught per trip did not show a clear trend during the study period. Minimum values were observed in November 2006 and 2007 (Fig. 2A), and maximum values were observed in October 2005; January, April, September, and December 2006; and July and October 2007. The overall mean CPUE for the study period was 3 individuals per trip. The distribution of numbers of fish caught per trip shows that ~85% of trips yielded 0–6 individuals, with 7–21 individuals caught during the remaining

¹ Mention of trade names or commercial companies is for identification purposes only and does not imply endorsement by the National Marine Fisheries Service, NOAA.

15% of trips. CPUE in excess of 13 individuals per trip occurred only between February and May.

CPUE in weight (kilograms) of fish caught per trip (Fig. 2B) showed a decreasing trend during 2006 but was much more stable in 2007, albeit with a decreasing trend during the first 6 months. The months with the minimum and maximum values of CPUE by weight were the same months for which the highest and lowest CPUE in numbers of fish per trip was observed. The overall mean CPUE for the study period was 133 kg per trip. The average weight of individuals caught during the 29-month sampling period was 41.9 kg, and peaks in weight occurred in the second and last quarters of each year of study.

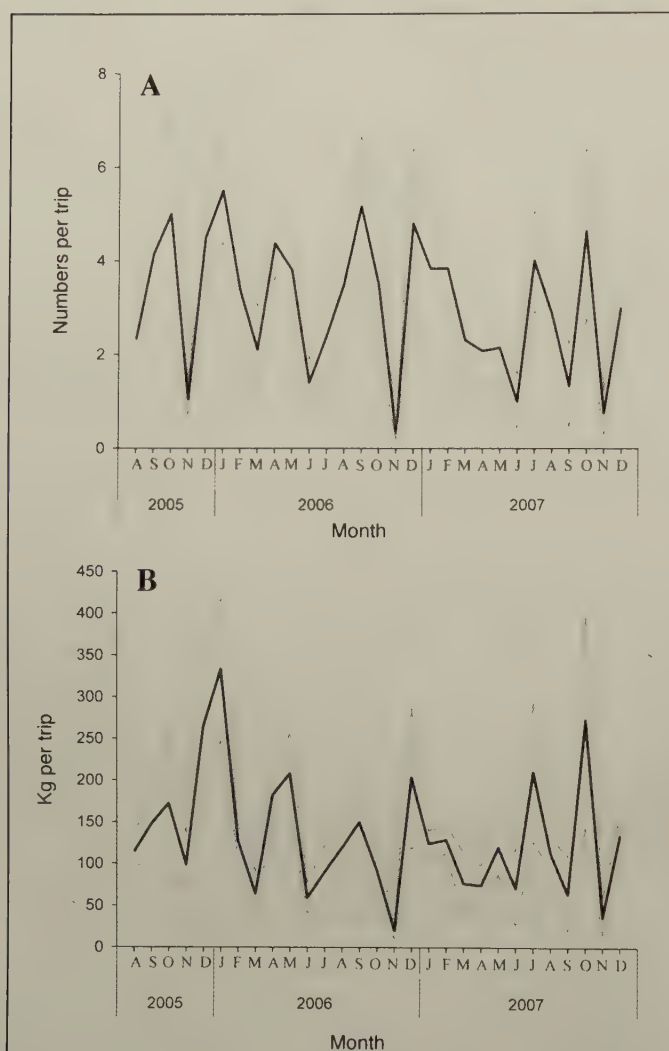


Figure 2

Catch per unit of effort of *Aetobatus narinari* in the Los Frailes Archipelago during the period from August 2005 to December 2007 in (A) numbers per trip and (B) weight (kg) per trip. Thick lines represent mean values, and thin lines are ± 1 standard errors.

Length structure and length-weight relationships

Measurements of length (DW in centimeters) were taken from 321 males and 846 females. Both sexes showed a unimodal length-frequency structure (Fig. 3A), and the overall range of observed lengths was 64–226 cm DW. The largest observed male was 190 cm DW. Males larger than 160 cm DW represented less than 10% of the sample. Females reached greater lengths, and the largest observed individual was a gravid female at 226 cm DW. Approximately 42% of females were larger than 160 cm DW.

Measurements of length and weight were taken from 105 males and 185 females. The comparison of male and female length-weight relationships showed no statistically significant differences ($F=1.87$; $P=0.16$). Therefore, a single length-weight relationship for both sexes was estimated (Fig. 3B). Point estimates and confidence intervals for intercept (a) and exponent (b) of this relationship were 1.824×10^{-5} (9.6×10^{-6} – 3.4×10^{-5}) and 2.95 (2.82–3.07), respectively.

Sex ratio and reproductive period

Of 321 males, 127 were immature and 194 were mature. Males were absent in the sampled catch during November 2005; August–December 2006; and June, September, and November 2007. Mature males were observed in all remaining months, except October 2005 (Table 1).

Of the 846 female individuals examined, 481 were immature, 242 were mature nongravid, 61 were gravid, and 62 were postgravid. Females were observed in different maturity stages in all months of the study period, except in November 2007, when all were found to be immature. November 2007 also was the month with the smallest sample size obtained during this study ($n=3$). Postgravid individuals, indicating recent parturition, were present in August 2005, February–May and July–September 2006, January–April and July–October 2007, and December 2007 (Table 1).

Fecundity and embryo lengths

Of the 61 gravid females, 75% had 3–5 embryos, and the remaining 25% of gravid females had 1–2 embryos. Seven individuals exhibited the maximum observed fecundity of 5 embryos within the uterus. Mean overall fecundity and standard deviation (SD) was estimated at 3.09 (SD=1.31) embryos. No significant relationship was observed between female length and the number of embryos for a sample of 35 gravid females ($F=2.29$; $P=0.14$).

The largest numbers of embryo samples were obtained in February 2006 ($n=13$), May 2006 ($n=8$), February 2007 ($n=22$), and July 2007 ($n=9$). Of 80 embryos observed, the length ranged from 10.1 to 44.5 cm DW (mean=31.5 cm DW) (Table 1). Eight embryos

with lengths of 10.1–11.5 cm DW still had their yolk sac; in larger specimens, the yolk sac had disappeared completely and nourishment had been provided by uterine milk or histotrophe through the numerous trophonemata present in the adult uterine walls.

Size at sexual maturity

The smallest recorded sizes at which females were nongravid, gravid, and postgravid were 106 cm, 150 cm, and 167 cm DW, respectively. The smallest mature male measured 97 cm DW.

The average length (L_{50}) at maturity was estimated at 129.2 cm DW for males (Fig. 4A). Joint confidence intervals (95%) for parameter estimates ranged from 125 to 134.9 cm DW for L_{50} and from 0.105 and 0.195 for the slope of the regression. For females, L_{50} was estimated at 134.9 cm DW (Fig. 4B), and joint confidence intervals ranged from 128.8 to 139.8 cm DW for L_{50} and from 0.09 to 0.216 for the slope.

Discussion

Effort, catch, and catch per unit of effort

Aetobatus narinari is usually captured as incidental catch in artisanal and industrial fisheries throughout its range. Directed fisheries for this species are uncommon and, to our knowledge, this study and that of Cuevas-Zimbrón et al. (2011) are the first accounts of fisheries for which *A. narinari* is the target species.

The total catch landed during the 29-month study period from August 2005 to December 2007 was 55.9 metric tons, 64% of which was taken during the first 6 months of 2006 and 2007. In this fishery, a close relationship was observed between catch in weight and numbers and fishing effort in number of trips (coefficient of determination [r^2]=0.85), but peak catches did not correspond with high CPUE values. Maximum observed catches in the first months of 2006 and 2007 were preceded by high CPUE values in the months of December 2005 and January and December 2006, during which 100% of the fishing trips were positive for catch of *A. narinari*. CPUE, especially in weight per trip, tended to decrease as the fishing season progressed, which may be explained by local depletion or migration (or by both) of *A. narinari* out of the fishing area studied. It is likely that the high CPUE values and percentage of positive trips in December and January acted as a trigger for initiating the fishing season in the first months of the year. The overall estimated mean CPUE of 3 individuals caught per trip in the Los Frailes Archipelago, Venezuela, during this study was similar to values reported for Seybaplaya, Mexico, by Cuevas-Zimbrón et al. (2011) but less than the 6 individuals per trip reported for Campeche, also in the southeastern Gulf of Mexico. However, trips in Seybaplaya and in the Los Frailes Archipelago were for 1 day, whereas

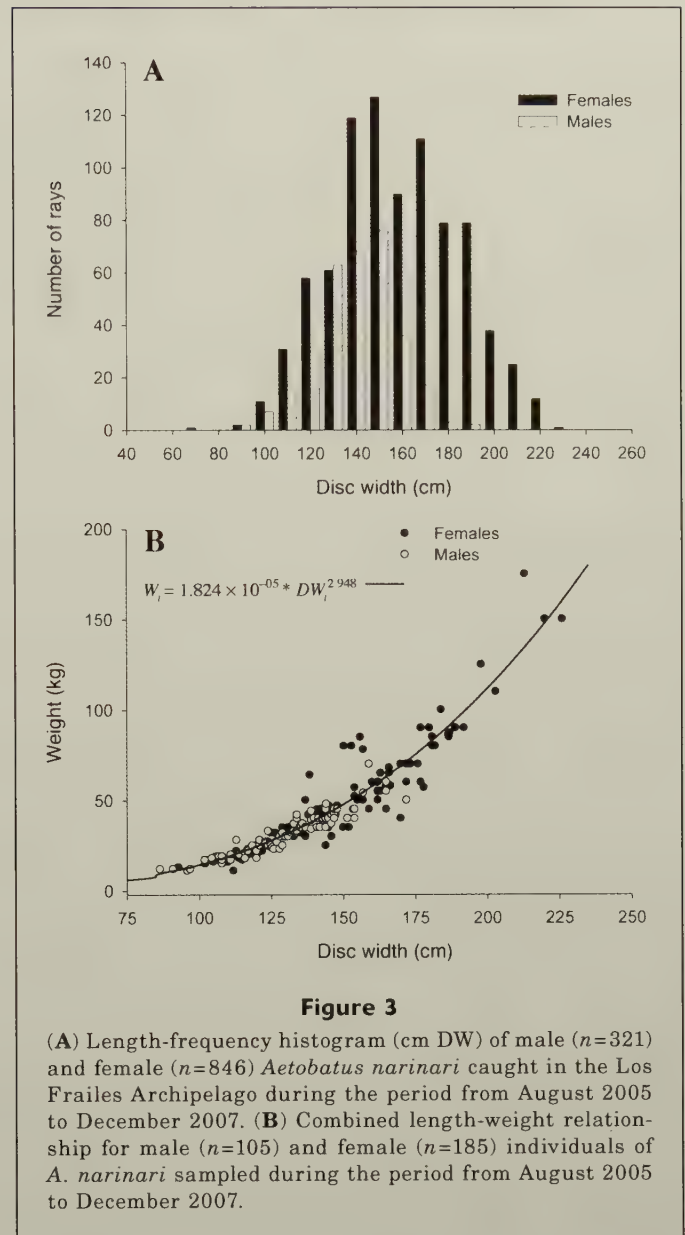


Figure 3

(A) Length-frequency histogram (cm DW) of male ($n=321$) and female ($n=846$) *Aetobatus narinari* caught in the Los Frailes Archipelago during the period from August 2005 to December 2007. (B) Combined length-weight relationship for male ($n=105$) and female ($n=185$) individuals of *A. narinari* sampled during the period from August 2005 to December 2007.

trips in Campeche were for 1–3 days. High numbers of fish caught per trip were observed in our study during December–February, coincident with periods of high availability of *A. narinari* reported for the southeastern Gulf of Mexico (Cuevas-Zimbrón et al., 2011). High CPUE values in our study also occurred in July–October, but they did not result in an important increase in fishing effort and catch. This lack of increase in fishing effort and catch is probably related to the existence of more lucrative alternate fisheries (*Scomberomorus* spp. and *Octopus* spp.) that are active particularly during this season of the year. A similar switch in target species (from *A. narinari* to *Octopus maya*) occurs during the second half of the year in the southeastern Gulf of Mexico (Cuevas-Zimbrón et al., 2011).

Table 1

Number of female and male spotted eagle ray (*Aetobatus narinari*) per maturity stage, number of female and male embryos, and mean length or disc width (DW) of embryos (cm) sampled per month between August 2005 and December 2007 in the Los Frailes Archipelago in northeastern Venezuela. Female maturity stages include nongravid (M1), gravid (M2), and postgravid (M3). Numbers in parentheses represent embryo size ranges (cm DW).

Year	Month	Females	Immature	M1	M2	M3	Males	Immature	Mature	Embryos	Females	Males	Mean DW (cm)
2005	A	29	18	5	2	4	6	2	4	2	2		40.5 (40.0–41.0)
	S	22	15	5	2		4	1	3	3		3	40.0 (39.7–40.6)
	O	19	17	1	1		2	2		4	2	2	40.3 (38.5–41.5)
	N	22	5	16	1								
	D	17	13	3	1		8	3	5	2	1	1	30.8 (21.5–40.0)
2006	J	22	10	9	3		8	2	6	3		3	38.8 (37.0–39.8)
	F	76	40	19	9	8	63	19	44	13	7	6	33.3 (11.5–44.5)
	M	12	8	2		2	8	5	3	1	1		41.0
	A	119	75	32	7	5	65	40	25	4	1	3	37.6 (36.5–40.0)
	M	90	46	35	6	3	20	8	12	8	5	3	27.9 (21.5–35.0)
	J	10	8	2			4	1	3				
	J	20	13	4	1	2	6	2	4	5	3	2	33.3 (26.0–40.0)
	A	28	4	20	3	1							
	S	23	14	4	3	2							
	O	11	8	2	1								
	N	4	3	1									
	D	21	16	4	1								
2007	J	53	39	11	2	1	20	6	14	3	2	1	28.2 (27.5–29.2)
	F	97	53	17	13	14	43	14	29	22	8	14	28.3 (10.8–39.0)
	M	16	15			1	14	6	8				
	A	11	7	3		1	15	5	10				
	M	11	4	6	1		1		1	1	1		17.8
	J	4	2	2									
	J	22	19	9	3	1	11	3	8	9	4	5	26.7 (10.1–36.0)
	A	39	22	16		1	10		10				
	S	4	1	2		1							
	O	26	2	10	1	13	9	6	3				
	N	3	3	—									
	D	15	11	2		2	4	2	2				
Total		846	481	242	61	62	321	127	194	80	37	43	

In our study of the small, directed fishery in north-eastern Venezuela, data required to analyze fishing effort by considering the effects of net size, effective fishing time, and other factors, such as depth and location, were not collected. Hence, effort was expressed in number of trips, which is likely a biased estimate of effective fishing effort. In any case, the time series analyzed is too short to infer changes in population abundance. Almost 40 years ago, Ginés et al. (1972) mentioned that the overall abundance of rays, including the spotted eagle ray, in northeastern Venezuela had decreased. Decreases in abundance of *A. narinari* also were mentioned for the Colombian Caribbean (Correa and Manjarrés, 2004); the northern Gulf of Mexico (Shepherd and Myers, 2005), where it was last observed in 1980 in autumn demersal trawl surveys; and Campeche Bank in the southeastern Gulf of Mexico (Cuevas-Zimbrón et al., 2011), as well as globally for the *A. narinari* species complex. Nevertheless, our study area is apparently part of an important concentration area for *A. narinari* in the Caribbean. This species may do better in this area than in other areas in the western Atlantic, because large populations of bivalve and gastropod mollusks, which constitute the main food items in the diet of *A. narinari* (see Randall, 1967), are present in northeastern Venezuela (Ginés et al., 1972; Lodeiros-Seijo and Freitas-Valbuena, 2008).

Length structure and length-weight relationships

In our study, females attained sizes larger than the sizes reached by males and were much more abundant than were males at lengths >160 cm DW. Differences in length distributions by sex have been reported in other areas of this species' range (Cuevas-Zimbrón et al., 2011). Additionally, growth studies on *A. narinari* indicate that females grow more slowly and reach larger sizes than do males (Dubick, 2000).

Males and females appear to be fully recruited to the fishery at 140 and 150 cm DW, respectively. Of *A. narinari* captured under 140 cm DW, ~37% were male and 19% were female. For both sexes, individuals <100 cm DW were rarely found in the fishery in the Los Frailes Archipelago. This absence may result from the selectivity of fishing gear or differential distribution of juveniles and adults. In the directed fishery in the southeastern Gulf of Mexico (Cuevas-Zimbrón et al., 2011), mesh openings (30.5–36.5 cm extended) are similar to the mesh sizes of nets used in northeastern Venezuela. However, despite the similar mesh openings used in both areas, *A. narinari* in the southeastern Gulf of Mexico, observed at lengths of 44–202 cm DW, included a higher proportion of juveniles than did the *A. narinari* observed in the Los Frailes Archipelago. Also, Cuevas-Zimbrón et al. (2011) reported size segregation in relation to distance from shore and depth in the southeastern Gulf of Mexico, with larger individuals predominating at distances of 30–50 km offshore

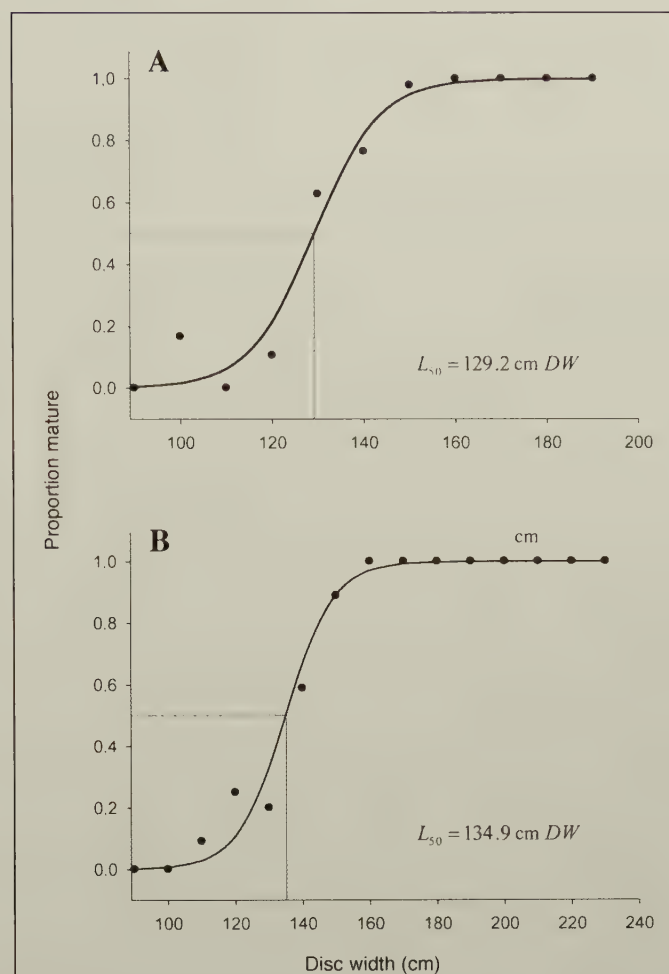


Figure 4

Length at maturity logistic functions for (A) male and (B) female individuals of *Aetobatus narinari* in the Los Frailes Archipelago during the period from August 2005 to December 2007. Horizontal and vertical lines indicate estimated length at 50% maturity (L_{50}) and value of L_{50} is given.

(depths of 8–12 m) and smaller individuals predominating at distances of 8–15 km offshore (depths of 6–8 m). In northeastern Brazil, neonates and juveniles of *A. narinari* were caught close to the shore in shallow depths <10 m (Yokota and Lessa, 2006). Considering these results by Cuevas-Zimbrón et al. (2011) and Yokota and Lessa (2006) and considering that the typical height (8–10 m) of nets used around the Los Frailes Archipelago precludes their use in shallower waters, it is likely that differential distribution of juveniles and adults in relation to the fishery in northeastern Venezuela explains the absence of small individuals in our samples.

There were no significant statistical differences in the length-weight relationships of male and female *A. narinari*. To our knowledge, our study is the first reported comparison of this relationship for this species.

Our estimate of the slope (b) for males and females combined is similar to the one ($b=3.13$) reported by Torres (1991) for species of the *A. narinari* complex in South African waters.

Sex ratio and reproductive period

In our study area, females were more abundant than males during most months of the period analyzed. Cuevas-Zimbrón et al. (2011) reported differences in sex ratios depending on depth and distance from shore in the southeastern Gulf of Mexico, where males were dominant in shallow waters close to shore and females were more abundant in deeper, more distant waters. Such spatial segregation by sex may explain the observed sex ratio patterns in our study. Additionally, sex ratios showed no significant differences in February, March, and June 2006 and March and April 2007; it is likely that these periods correspond to increased mating activities. Cuevas-Zimbrón et al. (2011) observed an increased proportion of adult females during March and April in the nearshore, shallow waters of Campeche Bank. These results indicate that migratory inshore-offshore movements relate to mating activity in adult *A. narinari*.

To our knowledge, this study is the first one to present *A. narinari* reproductive periodicity on an annual basis. Females in different maturity stages were found year round in the Los Frailes Archipelago. However, postgravid females were present in August 2005, February–May and July–September 2006, and January–April, July–October, and December 2007. Therefore, it appears that parturition occurs mainly during the periods of February–May and July–October. Similarly, Cervigón and Alcalá (1999) reported the presence of gravid females in March and April around the Los Roques Archipelago off central Venezuela. In India, gravid *A. ocellatus* females in “good number” were reported during April–May (Raje et al., 2007).

Schluessel et al. (2010b) observed mature oocytes and embryos in the same individual of *A. ocellatus*, and Uchida et al. (1990) reported that copulation followed immediately after parturition in aquarium conditions for *A. ocellatus*. For *A. narinari* in our study area, it is likely that mating occurs more intensely during February–May, considering the more balanced sex ratios and presence of postgravid females observed during this period.

Fecundity and embryo lengths

Fecundity of *A. narinari* has been reported by different authors to be 1–4 embryos (Gudger, 1914; McEachran and de Carvalho, 2002), a level similar to the fecundity observed in *A. ocellatus* (see Devadoss, 1984; Uchida et al., 1990). In our study, 75% of gravid females had 3–5 embryos, and the remaining 25% had 1–2 embryos. These minimum values may have been caused by abortions associated with stress during the capture process. However, captive *A. ocellatus* have been observed to give birth to only 1 or 2 pups (Uchida et al., 1990). From our

results, mean fecundity was 3.09 (SD=1.31) embryos per female. Additionally, no relationship was found between the length of gravid females and the number of embryos present. Gravid females of *A. narinari* had only one functional uterus in which all embryos were located—an observation also reported for *A. ocellatus* (see Schluessel et al., 2010b).

McEachran and de Carvalho (2002) indicated lengths at birth for *A. narinari* to be between 18 and 36 cm DW. In our study, the maximum embryonic length was 44.5 cm DW, and 40% of observed embryos were >36 cm DW. It is, therefore, likely that length at birth is >40 cm DW in our study area. This size is larger than the 30–40 cm DW reported for newborns in northeastern Brazil (Yokota and Lessa, 2006) but consistent with the 44 cm DW observed by Cuevas-Zimbrón et al. (2011) for neonates in the southeastern Gulf of Mexico.

Size at sexual maturity

To our knowledge, there has been only one previous report of size at sexual maturity for *A. narinari*. Dubick (2000) estimated that size at maturity was 122 cm DW for males and 124 cm DW for females in southwestern Puerto Rico. These results are slightly lower than the lengths obtained for males in our study, $L_{50}=129.2$ cm DW (95% confidence interval [CI]=125–134.9 cm DW), and lower than the results we obtained for females, $L_{50}=134.9$ cm DW (CI=128.8–139.8 cm DW). For *A. ocellatus* in the western Pacific and Indian Oceans, size at first maturity for males has been estimated at 99.8 cm DW in Indonesia (White and Dharmadi, 2007), 130 cm DW in Australia and Taiwan (Schluessel et al., 2010b), and 135 cm DW in Madras, India (Raje et al., 2007). Female size at first maturity has been reported for Australia and Taiwan at >150 cm DW (Schluessel et al., 2010b) and for India at ~150 cm DW (Raje et al., 2007). Several factors may determine variations in estimates of length at first maturity: true differences between populations, sample size, sampling bias, differences or errors in assigning maturity stages, and estimation methods. Because *Aetobatus* spp. are captured mainly throughout their range as bycatch in industrial and artisanal fisheries, the collection of adequate sample sizes has been a major limitation in studying these species. For example, White and Dharmadi (2007) and Schluessel et al. (2010b) studied only 28 and 55 male *A. ocellatus*, respectively. In the study by Schluessel et al. (2010b) only 1 of 56 female individuals was mature and the length at maturity was estimated at >150 cm DW. The directed nature of the fishery for *A. narinari* in northeastern Venezuela allowed us to obtain a much larger sample than in previous studies of length at maturity of *Aetobatus* spp.

Conclusions

Aetobatus narinari has been classified as near threatened in the IUCN Red List of Threatened Species. How-

ever, with the recent taxonomic changes within the species complex, the conservation status needs to be reviewed (White et al., 2010). For example, White et al. (2010) consider that *A. ocellatus* is more threatened than other members of this species complex, given that most threats listed for this complex have been from the Indo-Pacific region. Nevertheless, despite the paucity of data, there are indications in the western Atlantic that fishing has significantly affected the abundance of *A. narinari* (see Ginés et al., 1972; Claro et al., 2002; Correa and Manjarrés, 2004; Shepherd and Myers, 2005; Cuevas-Zimbrón et al., 2011). In particular, the small-scale, directed fisheries in the southern Gulf of Mexico and northeastern Venezuela that capture juvenile, mature, and pregnant individuals are a concern for the viability of these populations in areas where they are apparently still relatively abundant. An additional threat to *A. narinari*, which is associated with coral reefs, is habitat loss. Coral reef habitats in the western Atlantic have been declining over the most recent decades, with more than 75% of Caribbean reefs considered threatened (Burke et al., 2011).

In waters around Florida, the catch of *A. narinari* has been completely banned, but, to our knowledge, most countries of the western Atlantic have no specific regulations regarding the capture of *A. narinari*. In Venezuela, the fishery for *A. narinari* is unregulated and precautionary management measures may be necessary to assure population viability. In this study, we present results regarding length structure by sex, length-weight relationship, length at maturity, fecundity, size and sex ratio at birth, and reproductive periodicity—all of which represent important data for demographic modeling and stock assessment techniques that are required to develop management recommendations for the *A. narinari* fishery in northeastern Venezuela. However, more research is needed in this area, particularly regarding growth and mortality estimates, spatial and temporal changes in abundance, and migration patterns of *A. narinari*.

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Abstract—Rockfish (*Sebastes* spp.) biomass is difficult to assess with standard bottom trawl or acoustic surveys because of their propensity to aggregate near the seafloor in high-relief areas that are inaccessible to sampling by trawling. We compared the ability of a remotely operated vehicle (ROV), a modified bottom trawl, and a stereo drop camera system (SDC) to identify rockfish species and estimate their size composition. The ability to discriminate species was highest for the bottom trawl and lowest for the SDC. Mean lengths and size distributions varied among the gear types, although a larger number of length measurements could be collected with the bottom trawl and SDC than with the ROV. Dusky (*S. variabilis*), harlequin (*S. variegatus*), and northern rockfish (*S. polyspinis*), and Pacific ocean perch (*S. alutus*) were the species observed in greatest abundance. Only dusky and northern rockfish regularly occurred in trawlable areas, whereas these two species and many more occurred in untrawlable areas. The SDC was able to resolve the height of fish off the seafloor, and some of the rockfish species were observed only near the seafloor in the acoustic dead zone. This finding is important, in that fish found exclusively in the acoustic dead zone cannot be assessed acoustically. For these species, methods such as bottom trawls, long-lines, or optical surveys using line transect or area swept methods will be the only adequate means to estimate the abundance of these fishes. Our results suggest that the selection of appropriate methods for verifying targets will depend on the habitat types and species complexes to be examined.

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Estimating species and size composition of rockfishes to verify targets in acoustic surveys of untrawlable areas

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Rockfishes (*Sebastes* spp.) are a group of species with a predilection for high-relief, rocky habitats, where biomass estimation from traditional bottom-trawl survey data is difficult or impossible. However, many of these rockfishes also occur semipelagically, so that acoustic biomass assessments are possible (Wilkins, 1986; Demer et al., 2009; Ressler et al., 2009; Rooper et al., 2010). Acoustically estimating fish abundance requires accurate target verification of species composition and size distribution—verification that is typically achieved with midwater or bottom trawls. Because bottom trawling is hampered in high-relief areas, so too are acoustic abundance estimates from these habitats, owing to inadequate information describing species-specific abundance and size composition for fishes on or near the seafloor. Therefore, habitat-specific rockfish distribution patterns have the potential to affect the accuracy and precision of survey biomass estimates when traditional bottom trawl or combination acoustic-bottom trawl survey methods are used (Cordue, 2006).

Evidence suggests that untrawlable areas can support different species assemblages than those found in trawlable areas (Matthews and Rich-

ards, 1991; Jagielo et al., 2003; Zimmermann, 2003). Untrawlable areas can also have different size classes or abundances of the same species (Matthews, 1989; Stein et al., 1992; O'Connell and Carlile, 1993; Rooper et al., 2007). The primary species thought to inhabit untrawlable areas in high abundance in Alaska are northern rockfish (*Sebastes polyspinis*), dusky rockfish (*S. variabilis*), juvenile Pacific ocean perch (*S. alutus*), and black rockfish (*S. melanops*; Clausen and Heifetz, 2002; Rooper et al., 2007). Additionally, some rockfish species that occur in Alaska are rarely encountered in bottom trawl surveys (e.g., tiger rockfish [*S. nigrocinctus*]), possibly because of their preference for rough, rocky, and therefore untrawlable habitat. For these reasons, there is a clear need for alternative assessment methods to accurately and precisely estimate rockfish distribution and abundance over untrawlable areas, so that, in conjunction with similar estimates from trawlable areas, rockfish stock assessments can be improved.

Critical for an accurate acoustic assessment of rockfishes is determining the vertical distribution of species and sizes and their relation to the

seafloor. Some size classes of the population may occur exclusively near the bottom (<1 m), where they cannot be acoustically differentiated from the seafloor (Ona and Mitson, 1996; Rooper et al., 2010). Therefore, the ability to estimate the distance of fishes off the seafloor is important in determining which species and size classes are acoustically observable.

We evaluated the ability of gear types to discriminate species and size compositions of fish for the purpose of determining the best methods for target verification for acoustic surveys for rockfishes in untrawlable habitats. We compared the body lengths and species diversity of rockfishes from a modified bottom trawl with two optical methods—a remotely operated vehicle (ROV) and a stereo drop camera (SDC). For the SDC, the vertical distributions among species were compared. The proportion of rockfish species inhabiting trawlable and untrawlable areas was compared. We also compared the time and cost to employ each survey method in order to make recommendations for efficient and cost-effective methods for target verification in acoustic surveys.

Materials and methods

The research was conducted southwest of Kodiak Island, Alaska, at an offshore bank locally known as the “Snakehead Bank” from 3 to 12 October 2009 (Fig. 1). The continental shelf of the Gulf of Alaska near Kodiak has been shaped by past glacial and seismic activity and generally comprises sedimentary bedrock covered with glacially deposited sediments overlying most of the shelf (Hampton, 1983). Much of the shelf south of Kodiak Island is a series of flat underwater banks with deep troughs carved by glaciers that separate adjacent flat banks. The Snakehead Bank is a relatively small (~210 km²), shallow bank on the outer continental shelf that protrudes from the shelf and abuts the continental slope. At its shallowest point, the bank rises to within ~65 m of the surface and deeper water (>150 m) is found on the continental shelf to the north. Much deeper water (200–2000 m) is located on the continental slope to the south and east. The depths of the Snakehead Bank are inhabited by a distinct assemblage of continental shelf rockfishes that typically extends to about 180 m depth (Rooper, 2008). The Snakehead Bank has long been a productive area for commercial rockfish fisheries (Clausen and Heifetz, 2002; Hanselman et al., 2007), and Gulf of Alaska bottom trawl survey tows conducted at the Snakehead Bank often have high catches of northern rockfish and dusky rockfish (e.g., von Szalay et al., 2010).

The research was conducted aboard two vessels, the NOAA ship *Oscar Dyson* and a chartered commercial

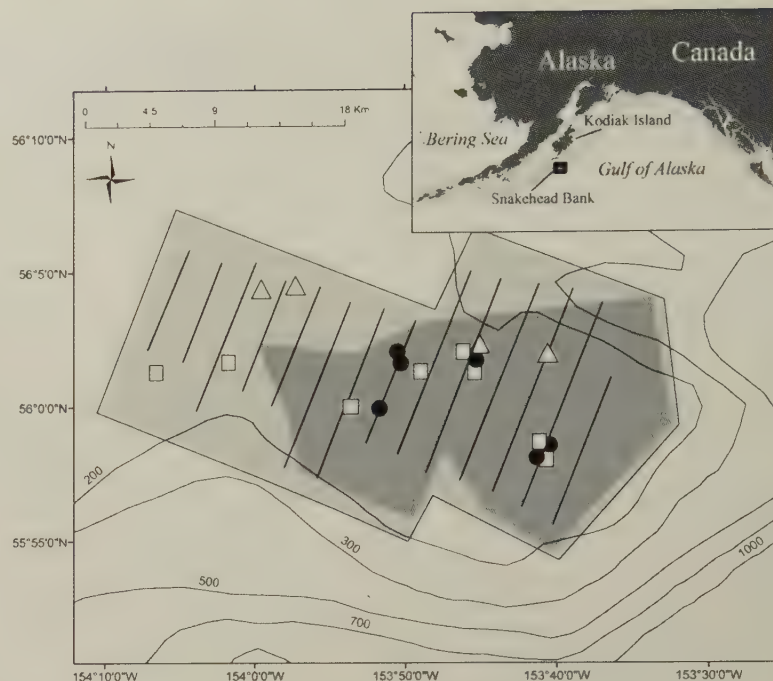


Figure 1

Map of study area showing the Snakehead Bank southwest of Kodiak Island, Alaska (indicated by black outline in inset). Deployment locations of remotely operated vehicle (triangles, $n=4$ deployments), stereo drop camera (squares, $n=8$ deployments), and bottom trawl (filled circles, $n=6$ deployments). Acoustic transect lines and depth contours (m) are also shown. Trawlable areas are shown in light gray and untrawlable areas are shown in dark gray.

fishing vessel, the FV *Epic Explorer*. The *Oscar Dyson* is a 64-m length overall stern trawler equipped for fisheries and oceanographic research. The *Epic Explorer* is a 39.6-m house-forward stern trawler active in commercial fisheries in Alaska. Both vessels were present in the study area simultaneously. Researchers aboard the *Oscar Dyson* collected acoustic data using Simrad EK60 scientific echosounders operating at five frequencies and a Simrad ME70 multibeam echosounder (Simrad, Horten, Norway¹). The ROV was also deployed from the *Oscar Dyson*. The modified bottom trawl and stereo drop camera were deployed from the *Epic Explorer*. During the survey with the *Oscar Dyson*, acoustic data were collected on a grid of parallel transects (Fig. 1). Eight individual passes of the parallel tracks were carried out (4 were completed during nighttime hours and 4 during daytime hours). From these acoustic data, researchers aboard the *Oscar Dyson* identified areas of fish aggregation and directed the deployment of the ROV, bottom trawl, and SDC to verify the species and length compositions of acoustic targets at those locations. Then the acoustic survey data were used to estimate abundance of fish species identified by the target verification meth-

¹Mention of trade names or commercial companies is for identification purposes only and does not imply endorsement by the National Marine Fisheries Service, NOAA.

ods (Jones et al., 2012 [this issue] for details of the acoustic assessment).

Remotely operated vehicle

Target verification was conducted with a Phantom DS4 ROV (Deep Ocean Engineering, Inc., San Jose, CA) nicknamed “*Sebastes*” that is owned and operated by the NOAA Southwest Fisheries Science Center (further details on this ROV and its capabilities can be found at <http://swfsc.noaa.gov/textblock.aspx?Division=FRD&id=8784>, accessed February 2012). Video footage from the ROV was recorded with a forward-looking color camera (Sony FCB-IX47C module with 468×720 lines of horizontal resolution and 18× optical zoom, Sony Corp., Tokyo, Japan). High-resolution still images were also collected with a Scorpio digital camera (Nikon Coolpix 995 with 4× zoom, Nikon Corp., Tokyo, Japan) to aid in species identifications. Speed of the ROV was measured by a downward facing Explorer Doppler velocity logger (DVL, Teledyne RD Instruments, San Diego, CA) which was also used to calculate transect length. This DVL was calibrated over a known distance and was accurate to ±0.07% (J. Butler, unpubl. data). The average speed of the ROV during deployments was 1.32 km/h (standard error [SE]=0.65) and the average altitude was 2.31 m (SE=0.75), although a constant speed, altitude, and heading was generally not maintained during deployment.

We used Canadian grid projections (Wakefield and Genin, 1987) calculated with 3-Beam software (Green Sky Imaging, Vero Beach, FL) to estimate the field of view for the ROV. This system uses 3 lasers on the ROV, the altitude of the vehicle and the pitch of the camera to calculate the field of view (Pinkard et al., 2005). The 3 high-intensity lasers were mounted parallel to the horizontal axis of the video camera: 2 parallel red lasers on either side of the video camera spaced 20 cm apart and 1 green laser that crosses the left parallel laser at 0.99 m and the right parallel laser at 2.72 m from the camera lens. The position of the green laser to the red lasers was used to calculate the distance from the camera lens to the seabed (i.e., slant range), and the parallel lasers provided a reference distance used to determine the field of view and fish length. For 3 of 4 transects with relatively flat seafloor, the field of view was calculated every 2 seconds. The average field of view, 2.61 m (SE=0.20), was used as an estimate of the search area for the remaining transect.

The ROV was deployed from the starboard side of the *Oscar Dyson* when weather permitted (Beaufort sea state <6) and was equipped with an acoustic transponder that provided its location relative to the ship. The position of the ROV on the sea floor was corrected in real-time by using WinFrog survey software (Fugro Pelagos, Inc., San Diego, CA). All other navigational data (e.g., water depth, temperature, heading, course over ground, etc.) were collected at 1–2 s intervals, synchronized, and logged by using WinFrog. The ROV

tether was attached with a swivel to a clump weight, which was connected by a cable to a winch onboard the vessel. The ROV and clump weight were lowered in unison to ~10 m above the seafloor at which point the cable to the clump weight was secured, monitored, and adjusted to maintain a clump-weight-elevation of >10 m (to avoid hitting the seafloor), while the ROV more closely approached the seafloor for identification of rockfishes and substrate type.

General locations for investigation were provided to the bridge from scientists operating the fisheries acoustics equipment and the ship's position was adjusted to drift or slowly navigate over a site where fish targets had been identified. However, the ROV did not transit specific transects and instead the seafloor was searched in one general direction, sometimes diverting from a straight-line to allow identification of rockfish targets or explore boulder patches more closely: this approach resulted in variable headings, speeds, and areas searched within a single deployment. For this reason, densities of rockfish were not computed from these transects. However, we did calculate the area swept by the ROV (distance traveled multiplied by the field of view) for comparison with the other gear types.

Bottom trawl

The bottom trawl used was a modified version of the Poly Nor'Eastern bottom trawl currently used by the Alaska Fisheries Science Center (AFSC) for bottom trawl surveys of the Gulf of Alaska and the Aleutian Islands (Britt and Martin, 2001; Stauffer, 2004). The net modifications included replacement of the standard footrope with rockhopper gear, the addition of heavier bridles (1.9 cm), and double meshes in the belly of the net. The center section of the rockhopper gear consisted of 61 cm rockhopper discs spaced approximately 46 cm apart. The rockhopper discs were spaced at about 61 cm on the wings and gradually tapered from 61 to 46 cm diameter on the wing extensions. All rockhopper discs were separated by solid sections of 2-cm (10-in.) discs. The bottom trawl was fished with 5-m² Fishbuster trawl doors each weighing 1089 kg (NET Systems Inc., Bainbridge Island, WA). The bottom trawl modifications were designed to improve the ruggedness of the net and allow the net to sample seafloor considered untrawlable with the standard survey net. The net width and height of the bottom trawl were ~17 m and 7 m respectively. The bottom trawl was towed at an average speed of 5.87 km/h (3.17 knots) ranging from 5.24 to 6.32 km/h and was generally deployed against the prevailing current. The area swept by the bottom trawl was estimated as the distance fished multiplied by the net width.

Stereo drop camera

The stereo drop camera system and deployment winch are described in Williams et al., (2010). The system consisted of two parallel-mounted cameras that collected simultaneous underwater video at a resolution

of 720×480 pixels. Each of the cameras was calibrated to correct for intrinsic optical parameters. Lengths of individual targets in the two cameras were calculated by identifying the position of individual points (such as a fish's head and tail) in each of the paired images and calculating their relative position using triangulation. The lens of each camera was keyed to its port so that the camera fit in only one position in the housing. This ensured consistent relative positioning of the cameras among deployments. Illumination was provided by two 50-watt, high-intensity discharge lights mounted above the camera housings inside an aluminum frame. The lighting system was powered by 4 rechargeable 4 Ah 12 V nickel-metal hydride batteries.

For calibration, the SDC was suspended in the water while the research vessel was dockside. The cameras were calibrated underwater by using images of a target plate with a printed 10×10 square checkerboard pattern of 50×50-mm squares (Williams et al., 2010). The approximate depth of the camera was 1 m and the approximate distance from the target was 1–2 m. The checkerboard target was lowered into the water along the vessel until it was plainly visible in both cameras. The target was then slowly moved horizontally and vertically through the field of view of both cameras and up to 15 minutes of calibration video were collected. Progressive scan video images were collected at 29.97 frames/s in each camera, and the videos from each camera were aligned by using a light-emitting diode (LED) synchronization light flashed in front of both cameras at the beginning of deployment. This LED synchronization was repeated at the end of the deployment to confirm that the video frames from the paired cameras were still aligned.

For the calibration procedure, still frame images were extracted from the aligned videos at 1-s intervals with Adobe Premiere software (Adobe Systems, Inc., San Jose, CA). Twenty paired images in which the target checkerboard was visible in both cameras were randomly selected for the calibration of the camera system. The calibration parameters were estimated with the camera calibration toolbox in Matlab (Mathworks, Inc., Natick, MA; Bouguet, 2008). For each image pair, the position of the corner points of the checkerboard pattern were identified by clicking on the images. The location of these points in the still images was computed by the calibration software to determine the focal parameters of each camera. Intrinsic camera parameters were used to correct the individual images for optical distortion resulting from the camera lenses.

The SDC was deployed and retrieved by an electric winch with 4-conductor electromechanical armored cable. The camera system was suspended 1–2 m off the seafloor at an angle of approximately 30° from horizontal to the seafloor. This position allowed a viewing path width of 2.43 m (SE=0.14) and under normal lighting conditions the field of view extended ~3 m in front of the SDC, although this varied with the distance of the SDC off the seafloor and the volume of light scattering particles in the water. The SDC traveled over the

seafloor at a target speed of 1.9–3.7 km/h (1–2 knots) for transects lasting up to about 1 hour. The overall mean speed of the SDC during field deployments was 2.26 km/h (SE=0.15). Some steerage of the camera was possible by towing the system gently with the vessel, and during slack water or low current periods the unit was sometimes towed to maintain a constant low speed. However, the direction of drifting and towing was with the prevailing current, and therefore directed transects were generally not possible. The area swept by the SDC was calculated as the path width multiplied by the distance traveled during a transect.

Classification of trawlable and untrawlable substrates

The substrata observed in the underwater video transects were classified by using the seafloor substrate classification scheme of Stein et al. (1992) and Yoklavich et al. (2000). It consists of a two-letter coding of substrate type denoting a primary substrate (>50% coverage of the seafloor bottom) and a possible secondary substrate (20–49% coverage of the seafloor bottom). In this classification scheme, there are seven substrate types: mud (M), sand (S), pebble (P, diameter <6.5 cm), cobble (C, 6.5 < diameter <25.5 cm), boulder (B, diameter >25.5 cm), exposed low-relief bedrock (R), and exposed high-relief bedrock and rock ridges (K). For example, a section of seafloor covered primarily in sand, but with boulders over more than 20% of the surface, would receive the substrate code sand-boulder (Sb), where the secondary substrate is indicated by the lower-case letter. Because the SDC and ROV provided a continuous display of substrata, the substrate code was only changed if a substrate encompassed more than 10 consecutive seconds of video.

For the purposes of this study, we further classified substrata as either untrawlable or trawlable with reference to the standard Poly-Nor'Eastern 4-seam bottom trawl used by the AFSC in biennial bottom trawl surveys of the Gulf of Alaska and Aleutian Islands (Stauffer, 2004). To define trawlability we used video captured from the ROV and SDC. The untrawlable areas were defined as any substrate containing boulders extending higher than ~20 cm off bottom or with exposed jagged bedrock that was rugose enough that the standard bottom trawl footrope would not pass easily over it. The heights of individual boulders and rocks were estimated by using the relative positions of the lasers from the ROV and measured with the SDC. The trawlable grounds, in contrast, were mostly composed of small cobble, pebble, sand, and mud without interspersed boulders or rocks. A single experienced observer conducted the substrate classification for both the ROV and SDC video transects.

Identification and measurements of fish

All rockfish caught with the bottom trawl were identified to species. Fish were identified and counted by species where possible for the optical methods (ROV and SDC). Fish were counted up to a maximum of 4 m

in front of the ROV and consistently out to 3 m in front of the SDC. *In situ* identification of fish with the optical methods was more difficult than with the bottom trawl and resulted in some fish that could not be positively identified to species. Many of these were smaller rockfish (<150 mm) that could not be positively identified to species with the ROV and SDC. Double counting of individual fish was assumed not to be an issue for the SDC because the camera drifted with the current in a relatively uniform direction and generally passed by fish as they were observed. For this vehicle, only fish that appeared in front of the camera were counted. In some cases during ROV deployments the vehicle was stopped so that an individual fish could be identified, and this brief pause could have resulted in double counting as fish milled around the stationary vehicle. An attempt was made to minimize double counting of individual fish in these cases by not counting fish that moved into the frame while the ROV was stationary; however, some double counting of fish probably occurred during these occasional stationary moments during ROV deployments. A single experienced observer identified the fish to species for both the SDC and ROV, and the habitat where each fish was observed was classified as either trawlable or untrawlable.

The Canadian grid projection (Wakefield and Genin, 1987) calculated with the 3-Beam software system was used to estimate fish length with the ROV. This limited the ability of the ROV to measure fish that were not in the same plane as the seafloor (i.e., above the seafloor). Additionally, the height of individual rockfish off the seafloor could not be measured.

For the bottom trawl, each catch was sorted to species and weighed. A random subsample of up to approximately 150 fish from each rockfish species identified in the catch was dissected to determine sex, and individual fork lengths were measured to the nearest centimeter. Because the bottom trawl integrates the catch spatially in both the vertical and horizontal planes, the height above the seafloor could also not be estimated for fish captured with the bottom trawl.

For the SDC, fish lengths were measured by using stereo triangulation functions supplied with the camera calibration software package (Bouguet, 2008) and the protocols identified in Williams et al. (2010). Images were extracted from the two video feeds at 1-s intervals, as with the calibration video. The videos from each camera were synchronized at the beginning and end by using the LED synchronization light. Length measurements were obtained by identifying the pixel coordinates of corresponding pixel locations (i.e., fish snout and fork of tail) in the left and right still frames of the camera. These points were used to solve for the 3-dimensional coordinates of the points in the images by triangulation, and by using the calibration-derived parameters. Once the 3-dimensional coordinates of the fish snout and tail were obtained, the length was measured as the simple Euclidian distance between the points in real space. This measurement method underestimated length for fish whose bodies were curved.

However, fish in the video and still camera rarely exhibited body curvature and the few individuals that did were excluded. All individual fish that could possibly be measured or a random sample of 200 fish per species (where more than 200 were possible) were measured for each deployment of the SDC.

For each fish that was measured with the SDC, the distance of the fish off bottom when it was first observed was also measured. These distances were then summarized into 0.5-m bins for each species. Because the SDC was deployed ~1 to 2 m off the seafloor, the vertical field of view was approximately 2 m off the seafloor and rarely extended above 3 m off the seafloor. This obviously limited the observed fish height off bottom.

Data analysis

Species diversity among the ROV, bottom trawl, and SDC samples was determined by examining the number of species observed with the 3 verification methods. The total number of species observed was compared among gear types by using analysis of variance (ANOVA) with video transects and bottom trawl hauls as replicates. The proportion of fish that were unidentified on each transect was also tested by using ANOVA to compare the ability of each of the gear types to allow identification of observed rockfishes to species. The proportion of unidentified fish by transect was the dependent variable for comparisons among the categorical variable of gear type. The proportion data were arcsin square-root-transformed before the tests to best approximate normality. Statistical significance for all tests was determined at $\alpha < 0.05$.

The fish-length distributions for major species were compared among gear types by using pairwise Kolmogorov-Smirnov tests to determine whether the length distributions from different gear types could have been drawn from the same sample. Fish-length composition was compared by using ANOVA to test for significant differences in mean length within major species for the 3 gear types. Owing to small overall sample sizes, individual fish lengths were used as replicates in this analysis and were combined across transects. The mean length of two rockfish species (northern rockfish and dusky rockfish) that occurred in both trawlable and untrawlable habitat were also compared to determine if fish were smaller in one habitat than in the other.

The percentage of rockfish that could be measured out of the total number of rockfish observed per transect for the major species was also calculated for each gear type. We used a *t*-test to determine whether the proportion of rockfish that could be measured was significantly different between the ROV and the SDC. For this analysis, the overall proportions of rockfish measured on each transect were used as the replicates. The proportion data were arcsin square-root-transformed before the *t*-test to improve normality. We did not consider this comparison for the rockfish captured in bottom trawl hauls because all the fish captured in the trawl could potentially be measured.

The acoustic dead zone is the area near the seafloor where fish targets cannot be resolved from the seafloor echo. At the Snakehead Bank, it was found to be depth dependent but generally extended to 0.7 m above the seafloor (Jones et al., 2012 [this issue]). Therefore, we calculated the proportion of each rockfish species that was observed in the acoustic dead zone (<1 m off the seafloor) and compared this proportion to a random vertical distribution of fish using a chi-squared statistic. This analysis was conducted only for fish whose height off the seafloor was measured with the SDC and was used to test the hypothesis that rockfish were randomly sorting themselves into heights off the seafloor, regardless of species.

The distribution of rockfish species between trawlable and untrawlable areas was also compared to a random distribution over the two habitats by using a chi-squared test. Additionally, the proportion of each of the major rockfish species and a combined "other" species group that occurred in untrawlable habitat was calculated along transects and compared to determine whether individual species were found in significantly different proportions in either trawlable or untrawlable habitats. For these analyses the replicates were transects where the species (or species group) occurred and where both trawlable and untrawlable areas occurred along that transect. Thus, the distribution of a rockfish species was tested as to whether it was found predominantly within trawlable or untrawlable habitat along a transect. The proportion data were arcsin square-root-transformed before the *t*-test to improve normality.

To produce a target verification map of backscatter from fish targets for acoustic analysis, we then assumed that the height of rockfish off the seafloor would have been the same for the fish observed in the ROV and captured in the bottom trawl (where this aspect of rockfish distribution was not measured) as was observed with the SDC. The proportions of each rockfish species <1 m off bottom and >1 m off bottom from the SDC were thus applied to the fish observed by the ROV and captured by the bottom trawl. The resulting proportions were shown graphically across the area of the acoustic survey where target verification transects and bottom trawl tows were conducted in order to show the spatial distribution of fish species, as well as their vertical distribution as either within or above the acoustic dead zone.

Finally, the amount of time needed to deploy and retrieve each gear type and process the data to completion was estimated. The amount of time for each task was summed by each gear type for comparisons. The approximate cost for building, deploying, and maintaining each of the gear types was also compared.

Results

Classification of substrate

The most common seafloor substrates observed in the ROV and SDC video data from the Snakehead Bank were

combinations of cobble, pebble, and sand. These 3 substrates comprised the primary substrate in 70.7% of the total seafloor area observed in the ROV videos and 89.8% of the seafloor observed in the SDC videos. However, 23.6% of these otherwise trawlable substrates observed in the ROV videos and 71.7% of these substrates in SDC videos were judged to be untrawlable because of the presence of large boulders or rocks. In total, 46.0% of the substrate observed by the ROV was designated as untrawlable, whereas 74.6% of the substrate observed by the SDC was designated as untrawlable. The untrawlable observations came predominantly from the eastern half of the study area. Acoustic data confirmed that the eastern half of the study area was mostly untrawlable and the western half of the bank was predominantly trawlable (Fig. 1; Weber et al.²). However, some patches of trawlable ground occurred at transects in the area designated as predominantly untrawlable and vice versa.

Identification of fish

The ROV was deployed at four locations, the bottom trawl was deployed at six locations, and the SDC was deployed at eight locations where acoustic backscatter attributed to fish was observed near the seafloor and in the water column (Fig. 1). During two of the SDC deployments only a single camera collected images and during one deployment at a trawlable location, no rockfish were observed. At 5 of the SDC sites, the bottom trawl was deployed at the same location immediately after SDC deployment. One of the ROV deployments was at the same location as that of a SDC deployment and two of the ROV deployments were at the same location as that of a bottom trawl (Fig. 1). However, all of the target verification deployments used in this analysis occurred between depths of 65 and 150 m on the top of the Snakehead Bank, and all were conducted within a 210-km² area.

Twelve different species of rockfishes were identified at the Snakehead Bank study area. Nine species were identified by using the ROV, 9 with the bottom trawl, and 7 with the SDC. Six species were observed in common by all 3 gear types. The most common rockfish captured in the bottom trawl and recorded by the ROV and SDC was dusky rockfish (Table 1). These were followed by harlequin rockfish (*S. variegatus*), northern rockfish, and Pacific ocean perch. Analysis of variance revealed there were no significant differences in the number of species observed among the three gear types ($P=0.31$, $F=1.27$, $n=16$ deployments). The total numbers of fish observed were almost equal for the ROV and SDC (1251 and 1176, respectively). The number of fish captured by the bottom trawl (6993) was much higher. The total amount of seafloor observed by the optical methods was

² Weber, T., C. N. Rooper, J. L. Butler, D. T. Jones, and C. D. Wilson. 2012. Seabed classification for trawlability using the Simrad ME70 multibeam echosounder on Snakehead Bank in the Gulf of Alaska. In review.

Table 1

Number of deployments, rockfish species observed or caught, percentage of rockfish not identified to species, total area swept, and percentage of area that was untrawlable for each gear type: remotely operated vehicle (ROV), modified bottom trawl (trawl), and stereo drop camera (SDC). Trawlability was defined in reference to the standard Poly-Nor'Eastern 4-seam bottom trawl used by the Alaska Fisheries Science Center in biennial bottom trawl surveys of the Gulf of Alaska and Aleutian Islands (Stauffer, 2004), not the modified bottom trawl used during our study.

		ROV	Trawl	SDC
Deployments		4	6	8
Rockfish observed				
Pacific ocean perch	<i>Sebastes alutus</i>	107	9	10
Dusky rockfish	<i>S. variabilis</i>	700	4733	500
Northern rockfish	<i>S. polyspinis</i>	31	254	148
Dark rockfish	<i>S. ciliatus</i>	7	40	8
Harlequin rockfish	<i>S. variegatus</i>	166	1942	151
Redbanded rockfish	<i>S. babcocki</i>	5		
Tiger rockfish	<i>S. nigrocinctus</i>	3		
Redstripe rockfish	<i>S. proriger</i>	80	2	
Pygmy rockfish	<i>S. wilsoni</i>		1	
Silvergrey rockfish	<i>S. brevispinis</i>		4	
Rosethorn rockfish	<i>S. helvomaculatus</i>			3
Yelloweye rockfish	<i>S. ruberrimus</i>	36	8	5
Unidentified rockfish	<i>Sebastes</i> spp.	116		351
Total rockfish species		9	9	7
Total rockfish observed		1251	6993	1176
Percentage unidentified		9.3%	0.0%	29.8%
Total area swept (ha)		2.70	4.66	2.62
Percentage untrawlable		46.0%	100.0%	74.6%

similar (~2.7 and 2.6 ha) and the amount of seafloor swept by the bottom trawl was much greater (4.7 ha).

There were significant differences in the percentages of fish identified to species with the 3 gear types by using ANOVA ($P=0.002$, $F=10.45$, $n=16$). The percentage of fish not identified to species was low for the ROV (9.3%), where control of the camera allowed individual fish to be followed and examined for species identification (Table 1). Fish identification was complete with the bottom trawl because all individuals could be closely examined and unambiguously identified. The high percentage of unidentified rockfish (29.8%) with the SDC reflects our inability to finely control the position and attitude of the drop camera system to closely examine fish for identification.

Measurement of fish length

Length distributions of dusky rockfish and harlequin rockfish were not significantly different ($P=0.71$ and $P=0.34$) between the ROV and SDC (Fig. 2). The length distributions were significantly different between the bottom trawl and the two optical methods (ROV and SDC) for dusky rockfish ($P=0.018$ and $P=0.013$) and for harlequin rockfish ($P=0.003$ and $P=0.002$). Length distributions for Pacific ocean perch were significantly

different ($P=0.03$) between the ROV and bottom trawl (there were not enough samples from the SDC to conduct statistical tests). Length distributions of northern rockfish from each of the gear types were significantly different ($P<0.01$).

Analysis of variance revealed that mean lengths of the major rockfish species collected in this study varied significantly among gear types (Fig. 3). Tukey's *post hoc* tests for 3 species of rockfishes (dusky rockfish, harlequin rockfish, and Pacific ocean perch) indicated there were no significant differences in mean length measured with the 2 optical gear types ($P>0.05$). Tukey's *post hoc* tests indicated that the mean length of northern rockfish from the ROV was significantly shorter than those estimated by the SDC, and northern rockfish measured by both the optical methods were significantly shorter than those measured from the bottom trawl. Mean lengths of harlequin rockfish from the ROV and SDC were significantly shorter than those from the trawl. Dusky rockfish and Pacific ocean perch mean lengths were the same for all 3 methods. In general, more shorter fish were observed with the optical methods than with the bottom trawl. Interestingly, the mean length of northern rockfish from untrawlable areas was shorter than that from trawlable areas (Fig. 4), although no differences in length by habitat (trawlable

or untrawlable) were observed for dusky rockfish. Because of the confounding of gear types for northern rockfish (all the small northern rockfish were measured by using the ROV in untrawlable areas and only one northern rockfish was measured in an untrawlable area with the SDC), these differences could not be tested for statistical significance.

The percentages rockfish observed on a transect that could be measured varied between the ROV and SDC, although this difference was not statistically sig-

nificant when a t -test was applied ($P=0.056$, $t=-2.3$, $df=7$). For the ROV an average of 9.9% ($SE=0.054$) of the dusky rockfish, northern rockfish, harlequin rockfish, and Pacific ocean perch observed on a transect could be measured. On average 41.9% ($SE=0.184$) of these species captured in a trawl haul were measured, higher than the percentage with the optic methods (Fig. 5). With the SDC, 35.6% ($SE=0.100$) of the rockfish species observed on a transect could be measured (Fig. 5).

Vertical distribution of fish and comparisons between trawlable and untrawlable areas

The results of the acoustic survey indicated that the majority of rockfish were near the seafloor because the mean height off bottom of rockfish from all 8 acoustic survey passes was 1.5 m (Jones et al., 2012 [this issue]). Mean height off bottom during each of the 8 survey passes ranged from 1 to 3.25 m, a range that allowed most of the rockfish biomass to be observed with the ROV or SDC or captured in the trawl. The observed height off the seafloor, as measured with the SDC, varied significantly among rockfish species from a random distribution according to a chi-squared test (Table 2). Harlequin rockfish, Pacific ocean perch, rosethorn rockfish (*S. helvomaculatus*), dark rockfish (*S. ciliatus*), and yelloweye rockfish (*S. ruberrimus*) were observed exclusively within 1 m of the seafloor (Fig. 6). The rockfish species found in the water column (>1 m off the seafloor) were dusky and northern rockfish, although these species were also found within the acoustic dead zone as well. The bottom trawl integrated rockfish catch from approximately 0 m to 7 m (the height of the net opening) off the seafloor and the ROV laser system does not allow for measurement of distance off the seafloor on a fine scale; therefore the depth distributions of various rockfish species could not be precisely determined with these gear types.

With a chi-squared test, we also detected a significant nonrandom distribution of rockfish species by habitat type; either trawlable or untrawlable (Table 3). The proportion of fish in untrawlable areas was higher than in trawlable areas for the individual fish species (Fig. 7), as well as for the combined other rockfish group (yelloweye rockfish, redstripe rockfish (*S. proriger*), redbanded rockfish (*S. babcocki*), dark rockfish, tiger rockfish, and rosethorn rockfish). T -tests indicated some of these differences were insignificant, because the proportion of dusky rockfish ($P=0.10$, $t=-1.83$, $n=10$), northern rockfish ($P=0.33$, $t=-1.07$, $n=6$), and Pacific ocean perch ($P=0.07$, $t=-2.12$, $n=8$) was not significantly higher in untrawlable areas than in trawlable areas. All the harlequin rockfish and the

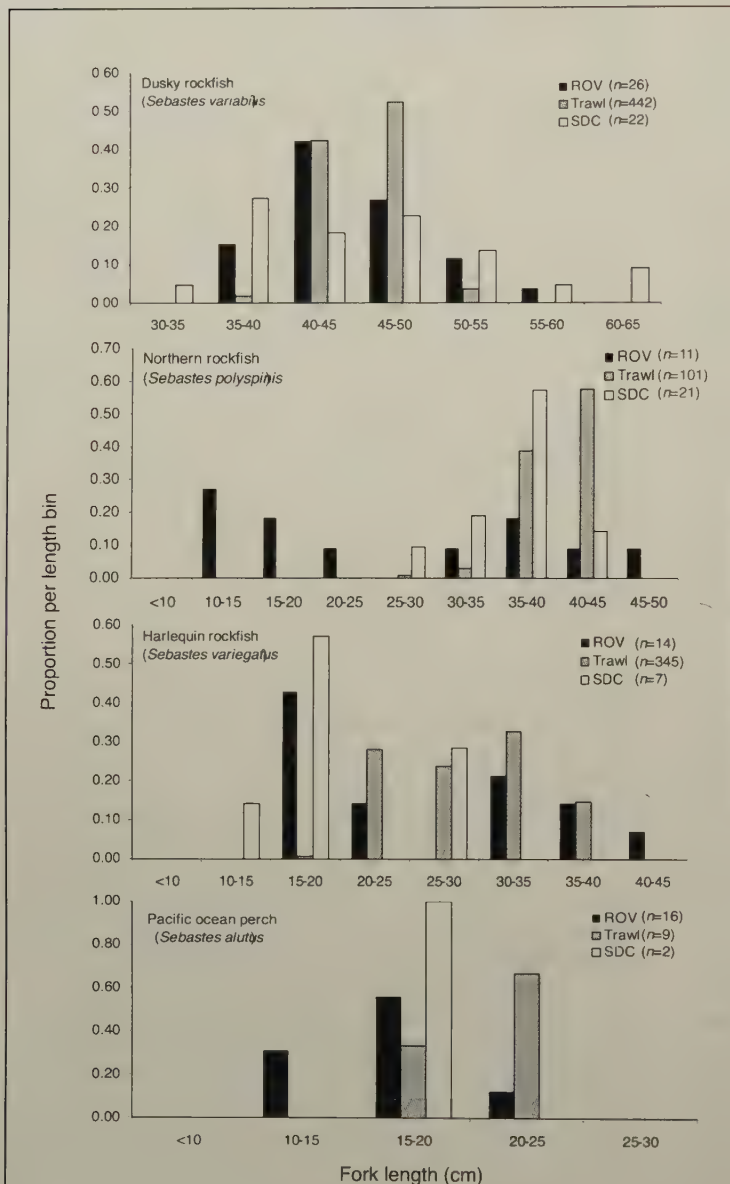


Figure 2

Length-frequency data for each gear type (ROV=remotely operated vehicle, Trawl=bottom trawl=Trawl, and SDC=stereo drop camera) for dominant rockfish species observed at the Snakehead Bank, Alaska, in 2009. n =the number of fish measured for each species and gear type.

rockfish grouped into the “other species” category were observed exclusively in untrawlable areas, with the exception of one redbanded rockfish (Fig. 7). This division resulted in a significantly higher proportion of the “other species” group being found in untrawlable areas than in trawlable areas ($P < 0.0001$, $t = -40.09$, $n = 12$).

Together, the differences in both vertical (height off the seafloor) and spatial (trawlable versus untrawlable habitat) distributions of the rockfish, resulted in a complex picture of the verification of fish species potentially observed in acoustic data during the survey of Snakehead Bank (Fig. 8). Rockfishes within the acoustic dead zone (<1 m) over trawlable areas were dominated by dusky rockfish and northern rockfish (Fig. 8). In the untrawlable areas, the acoustic dead zone contained dusky, harlequin, and northern rockfishes in greatest abundance. Fish in the water column (>1 m off bottom) that were likely to be observed by using the vessel acoustics comprised mostly dusky and northern rockfish in both trawlable and untrawlable areas, although as shown in Figure 7, the higher proportion of these two species was observed in untrawlable areas.

Data analysis, processing time, and cost

The ROV required both the highest level of expertise and the longest time to deploy (Table 4). The bottom trawl required the least amount of time to deploy, retrieve, and process samples (Table 4). The level of expertise required to deploy and retrieve the gear was high, but other tasks associated with the bottom trawl required moderate expertise. The level of expertise required to deploy and retrieve the SDC was also high, although it could be done in relatively short time. The level of expertise to process the SDC video footage into data required for acoustic surveys was also high, and the time required to collect and process one sample (1 h of video) was large (7 h). Once the ROV video was collected, processing it into data required for verification of target species in the acoustic surveys was comparable to that required with the SDC, although more time was necessary to measure the lengths of fish with the lasers than with the stereo cameras. The initial costs of purchasing the ROV and constructing the bottom trawl were quite high. The SDC was the cheapest of the 3 equipments to purchase and construct. The cost per unit of area surveyed during this project was cheapest with the bottom trawl and most expensive with the ROV.

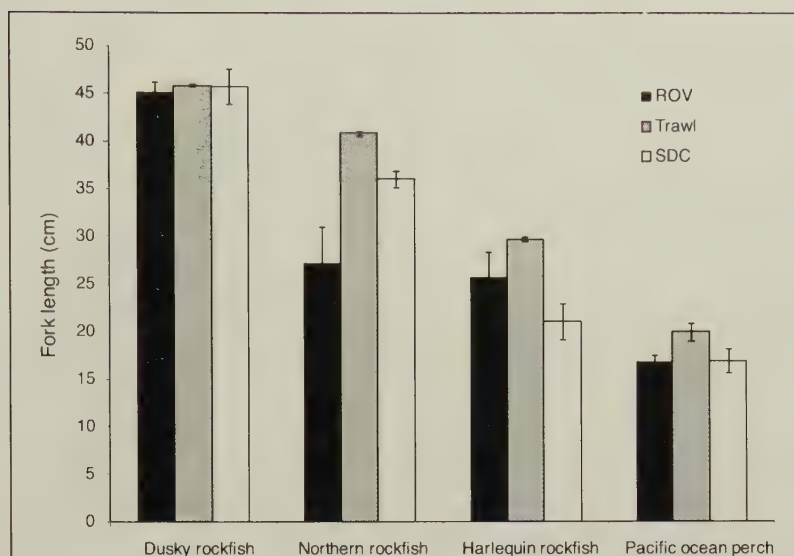


Figure 3

Mean (and standard error) fork length (cm) for dominant rockfish species observed with the 3 gear types (ROV=remotely operated vehicle, Trawl=bottom trawl, and SDC=stereo drop camera) at the Snakehead Bank, Alaska, in 2009. Sample sizes for length measurements are the same as those shown in Figure 2.

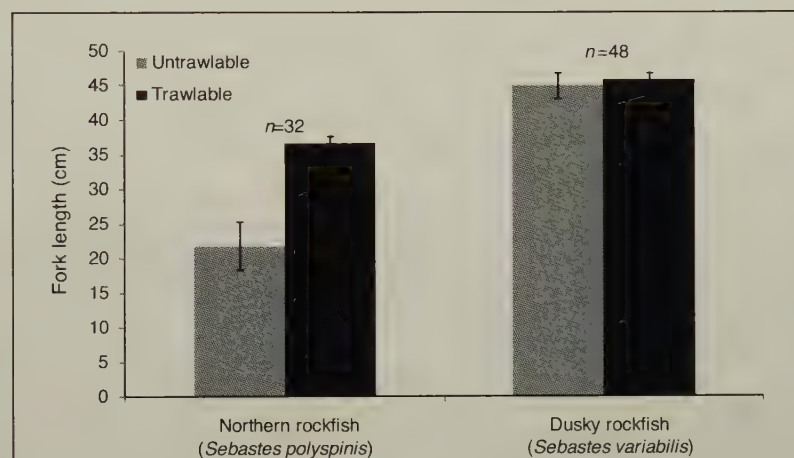


Figure 4

Mean (and standard error) fork length (cm) for rockfish species observed and measured in both trawlable and untrawlable regions of the Snakehead Bank, Alaska in 2009. Data from the remotely operated vehicle and stereo drop camera are combined. n =number of fish measured for each species.

Discussion

In this study, the rockfish species observed in the water column were similar between trawlable and untrawlable areas, which is encouraging for the potential to assess the biomass of these species acoustically in both types of habitats. However, clear differences

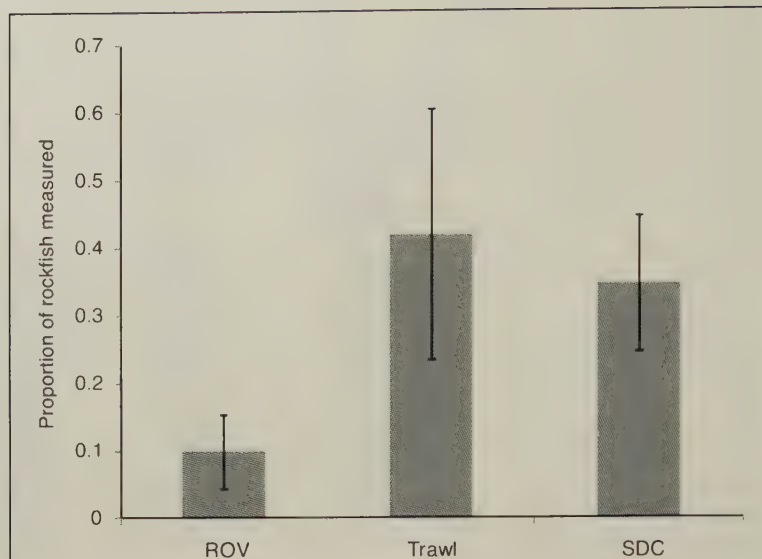


Figure 5

Average proportion (and standard error) of rockfish that were measured out of the total number of dusky rockfish (*Sebastes variabilis*), northern rockfish (*S. polyspinis*), harlequin rockfish (*S. variegatus*), and Pacific ocean perch (*S. alutus*) observed. Proportions were calculated for the 4 species from each transect surveyed with the remotely operated vehicle (ROV; $n=4$ deployments), bottom trawl haul (trawl; $n=6$ deployments) and stereo drop camera transect (SDC, $n=5$ deployments) conducted at the Snakehead Bank, Alaska, in 2009 where length data were collected. The average proportions were computed by using transects as replicates.

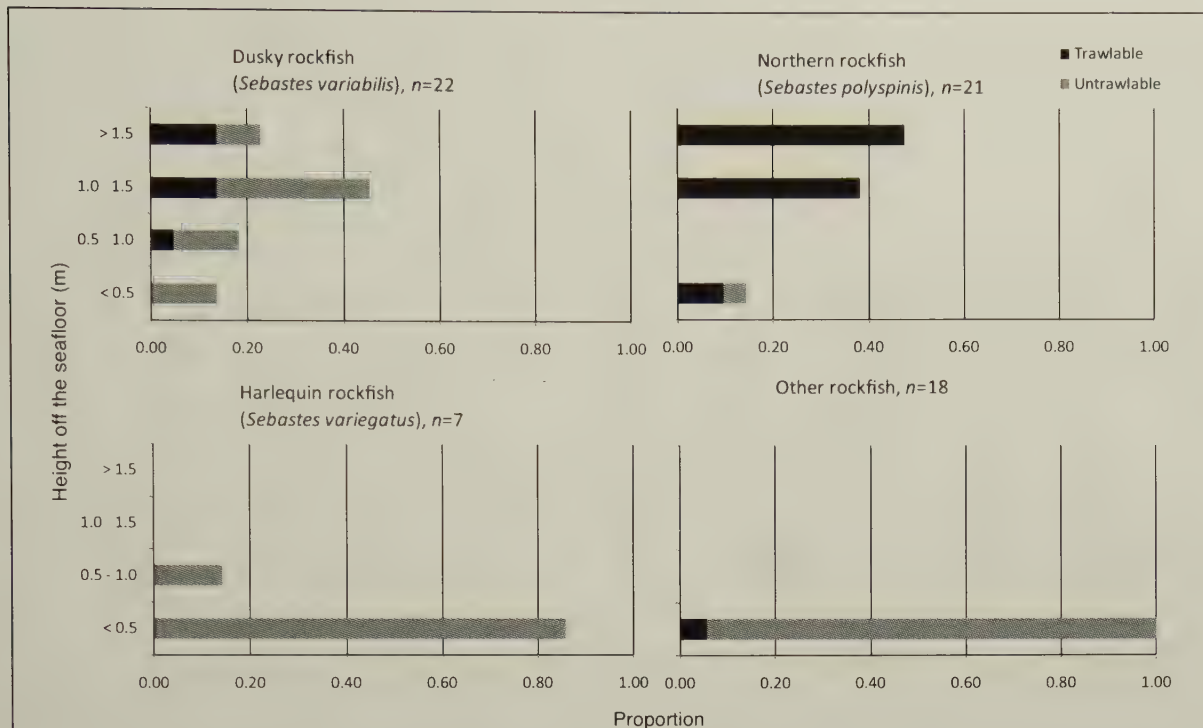
Table 2

Chi-squared test for random distribution of each rockfish species at <1 m height off the seafloor. The observed frequency and expected frequency of each rockfish species <1 m off the seafloor are shown for data from stereo drop camera deployments where both cameras were functional and rockfish were observed during the deployment ($n=5$ deployments).

Species	Observed frequency <1 m off bottom Species	Expected frequency <1 m off bottom
Dusky rockfish (<i>Sebastes variabilis</i>)	7	11
Northern rockfish (<i>Sebastes polyspinis</i>)	3	11
Harlequin rockfish (<i>Sebastes variegatus</i>)	7	4
Other rockfish: Pacific ocean perch (<i>Sebastes alutus</i>), rosethorn (<i>S. helvomaculatus</i>), yelloweye (<i>S. ruberrimus</i>), and dark rockfishes (<i>S. ciliatus</i>)	18	9
Total number of fish observed/ χ^2	68	19
χ^2 (critical value, $P=0.05$, $df=4$)		9.49

in rockfish species composition on the seafloor in trawlable and untrawlable areas were observed during this study. Other studies of untrawlable habitats have revealed similar differences in rockfish species composition near the seafloor when compared with trawlable areas (Matthews and Richards, 1991; Matthews, 1989; Rooper et al., 2007). Our observations highlight the potential that a considerable proportion of the rock-

fish biomass (in this case harlequin, northern, and dusky rockfish) will be unavailable to the standard bottom trawl survey in untrawlable areas, potentially negatively biasing population abundance estimates. Although at least some of these species may be available for acoustic biomass estimation, the abundance of species that are found in the acoustic dead zone in untrawlable areas will be more difficult to estimate

**Figure 6**

Distribution of rockfish species by height off the seafloor (m) at the Snakehead Bank, Alaska, in 2009. These data were available only from the five stereo drop camera transects where both cameras were functional and where rockfish species were observed. The data for each depth and species are split into trawlable and untrawlable proportions based on the seafloor characteristics where the individual fish were observed. Other rockfish include Pacific ocean perch (*Sebastes alutus*), rosethorn rockfish (*S. helvomaculatus*), yelloweye rockfish (*S. ruberrimus*), and dark rockfish (*S. ciliatus*). n=no. of fish in sample.

Table 3

Chi-squared test for the random distribution of rockfish species between trawlable and untrawlable habitats. Data from stereo drop camera (SDC) and remotely operated vehicle (ROV) deployments (n=12). Shown are the observed frequency and expected frequency of rockfish for each species that occurred in trawlable areas, based on the amount of trawlable area surveyed with the SDC and ROV are shown.

Species	Observed frequency in trawlable areas	Expected frequency in trawlable areas
Dusky rockfish (<i>Sebastes variabilis</i>)	157	479
Northern rockfish (<i>Sebastes polyspinis</i>)	130	71
Harlequin rockfish (<i>Sebastes variegatus</i>)	0	127
Pacific ocean perch (<i>Sebastes alutus</i>)	1	46
Other rockfish: yelloweye (<i>Sebastes ruberrimus</i>), redstripe (<i>S. proriger</i>) redbanded (<i>S. babcocki</i>), dark (<i>S. ciliatus</i>), tiger (<i>S. nigrocinctus</i>), and rosethorn rockfishes (<i>S. helvomaculatus</i>)	1	59
Number of fish observed/ χ^2	1960	366
χ^2 (critical value, $P=0.05$, $df=11$)		19.68

because these species are unavailable to both acoustic and bottom trawl surveys.

Temporal and spatial variability in species distribution may have influenced the results of comparisons

of species distribution by gear types in this study. Although each of the gear types was deployed at a slightly different combination of sites over the same relatively small area of the Snakehead Bank, each gear type was

deployed in reasonably close proximity in space over the same time period. We observed uniformity of species composition within trawlable and untrawlable habitats when sampling with the two optical gear types. For example, dusky rockfish and northern rockfish were the

dominant species observed with both optical gear types in trawlable areas, whereas additional rockfish species such as harlequin rockfish were found with the optical gear types (as well as the trawl) in untrawlable areas. This result would not be expected if we were sampling substantially different communities in the small area the Snakehead Bank. The acoustic information showed that the biomass of fish in the Snakehead Bank area was relatively stable between eight successive day and night passes (~2800 t, coefficient of variation [CV]=0.27; Jones et al., 2012 [this issue]), indicating it was unlikely that substantial fish movement into or out of the study area would have influenced the results.

The spatial scale of the effort varied also with each gear type in this study. The bottom trawl covered a wide area, whereas the two optical technologies covered only small swaths of the seafloor. This difference in spatial scale probably affected the catchability of the gear types. The substrate type also affected the catchability. In the more rugose substrate, the ROV and SDC allowed rockfish to be observed in individual cracks and crevices although identifying individuals partially hidden in crevices was more difficult with the SDC. The modified bottom trawl undoubtedly did not capture all the fish species that occurred in the most rugose areas. The modifications to the footrope were designed to allow the net to bounce over large rocks and probably led to some fish in rocky areas not being captured.

Fish length differed between the 3 gear types. The smallest fish were observed only with the

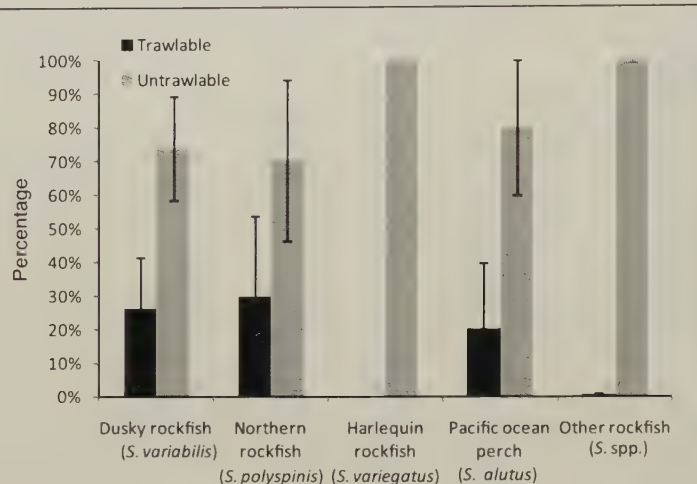


Figure 7

Mean percentage (and standard errors) of rockfish by species observed in trawlable and untrawlable areas as estimated with the stereo drop camera and remotely operated vehicle along the transects. Only transects that included both trawlable and untrawlable areas and transects where the rockfish species occurred were used to calculate the mean percentages. The other rockfish species group includes yelloweye rockfish (*Sebastes ruberrimus*), redstripe rockfish (*S. proriger*), redbanded rockfish (*S. babcocki*), dark rockfish (*S. ciliatus*), tiger rockfish, (*S. nigrocinctus*), and rosethorn rockfish (*S. helvomaculatus*).

Table 4

Estimated total hours spent deploying each gear type to collect one bottom trawl sample from a 5–15 minute tow and one hour of underwater images from either the remotely operated vehicle (ROV) or the stereo drop camera (SDC). Each major data collection task (identifying, counting, and measuring all species of fish) and data entry is listed, as well as the relative level of expertise required to complete the task. Underwater video was used for substrate classification, fish counting, and identification for the ROV and SDC. Still images were used for determining fish length for the ROV and SDC, and still images were used to aid fish identification with the ROV.

Task	ROV		Bottom trawl		SDC	
	Estimated time required (person hours)	Level of expertise required	Estimated time required (person hours)	Level of expertise required	Estimated time (person hours)	Level of expertise required
Deployment and retrieval of gear	2.5	High	1	High	1.5	High
Classification of substrate	0.5	High	—	—	0.5	High
Fish count and identification	2	High	0.5	Medium	2	High
Determination of fish length	3	High	0.5	Low	2	High
Data entry and formatting	1	Medium	0.5	Medium	1	Medium
Total	9		2.5		7	
Initial cost of equipment	>\$100,000		\$66,000		\$18,308	
Operational cost (per ha of seafloor)	\$1,393		\$139		\$262	

optical gear types and therefore they were either unavailable to or not retained by the bottom trawl; the smallest rockfish (<150 mm) could probably escape more readily through the bottom trawl. Escapement of this kind has been observed in the Gulf of Alaska and Aleutian Islands bottom trawl surveys where the smallest (<100 mm) rockfish are often not captured (von Szalay et al., 2010, 2011). Because juvenile fish and smaller species, such as harlequin rockfish were observed primarily in untrawlable areas, it is also possible that the smallest fish seek out shelter among the rocks and are not available to the bottom trawl. The accuracy of the fish length measurements also differed by gear type. The fish captured in the bottom trawl are generally assumed to be measured with only a minimal amount of error because each measured fish is individually handled, measured, and recorded. However, fish caught by bottom trawl are only measured to the nearest cm. The error rates for the SDC in measuring the size of known targets have previously been estimated to be less than 8.2%, or less than 2.5 cm for a 30-cm fish (Williams et al., 2010). Other stereo video systems have generally produced smaller error rates <1% of length (Harvey et al., 2002, 2003; Shortis et al., 2009). The higher error rates for the SDC are probably due to the need to remove cameras from the housing unit after each deployment, which possibly causes a slight misalignment of the cameras in relation to the position at calibration reducing the precision of measurements (Williams et al., 2010). The accuracy of length measurements from the ROV parallel laser measuring system was not determined; however, previous research with parallel laser systems have indicated length measurements are accurate to 1–5% of the total length of a rigid object (Rochet et al., 2006). Because fish lengths are translated directly into target strength estimates for acoustic biomass estimation, errors and biases in fish length from the target verification tools are important to determine so that the effect on total fish biomass can be known.

Thus, the results of this study indicate that the method chosen for target verification in acoustic assessments depends on the fish species to be assessed, their size, and the substrate type to be examined. Advantages of the bottom trawl over the optical methods are that it allows identification and measurements of all the rockfish species collected. Specimens collected with the bottom trawl also provide auxiliary information important to stock assessment, such as diet, age, and stage of maturity. The advantage of the optical methods is that they provide data for discriminating species assemblages in untrawlable areas or areas with potentially vulnerable habitats such as deepwater corals and sponges that could be damaged by further trawling (Heifetz et al., 2009). Habitat-specific densities

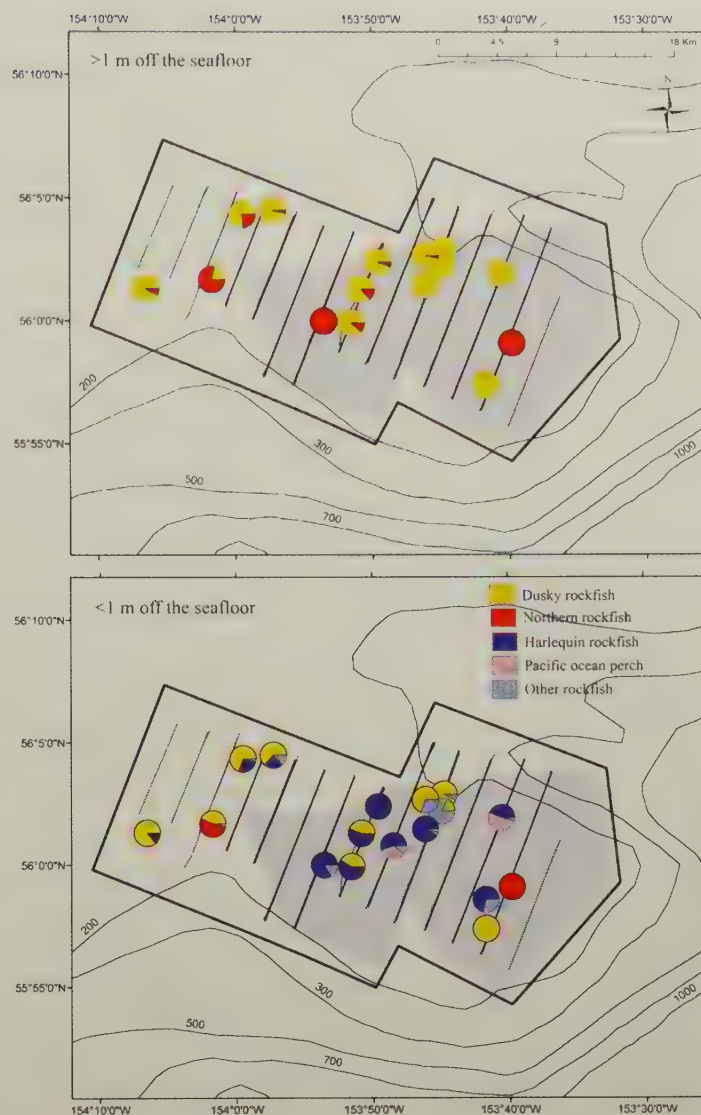


Figure 8

Composition of rockfish (based on percentages from stereo drop camera estimates of height distribution from each species) at stations by height off the seafloor in the categories >1 m off the seafloor and <1 m off the seafloor. Stations were surveyed by remotely operated vehicle, bottom trawl, and stereo drop camera. Some sites have been slightly offset to show species composition charts. Solid line indicates the extent of the acoustic transects, the shaded area shows the area that was considered predominantly untrawlable in the analysis of acoustics (Weber et al.²). Dusky rockfish (*Sebastes variabilis*), northern rockfish (*S. polyspinis*), harlequin rockfish (*S. variegatus*), and Pacific ocean perch (*S. alutus*) are shown individually. The other rockfish species group comprises yelloweye rockfish (*Sebastes ruberrimus*), redstripe rockfish (*S. proriger*), redbanded rockfish (*S. babcocki*), dark rockfish (*S. ciliatus*), tiger rockfish (*S. nigrocinctus*), pygmy rockfish (*S. wilsoni*), silvergrey rockfish (*S. brevispinis*), and rosethorn rockfish (*S. helvomaculatus*).

and associations can also be collected by video methods—factors that are masked by the bottom trawl that integrates the catch over a large and unobserved area

of the seafloor. Optical methods also allow researchers to collect length information from smaller individuals, but this advantage can be offset by potential inaccuracies in species identification because these small individuals are difficult to identify with optical methods.

There are cost advantages of using the SDC over both the ROV and trawl methods because the initial investment in equipment is smaller. The stereo cameras allow scientists to accurately measure the height of individual fish off the seafloor and the opportunity to measure the length of a higher proportion of observed fish than does the ROV. These are both critical factors for acoustic surveys where it is important to know the size of fish that are observed acoustically in the water column. The major disadvantages of the SDC are the difficulties associated with identifying all fish to species and an inability to finely control the position of the cameras.

For this analysis, we assumed that the distribution of height off bottom for each species was accurately represented by the data collected with the SDC. Any behavioral reactions to this camera system (for example fish diving away from the camera as it approached) would have influenced our ability to perceive the height of fish off the bottom accurately. Errors in this measurement would have serious effects on the acoustic estimates of abundance for any species that reacted to the SDC. For example, if one rockfish species had a tendency to dive to the seafloor before coming into the view of the SDC, as has been observed with manned submersibles (Krieger and Ito, 1999), the species could be under-represented in the biomass estimate of fish from above the acoustic dead-zone. As the SDC is a relatively small vehicle without a motor that drifts at low speeds with the prevailing current (creating less noise), its potential for eliciting a reaction by fishes is probably less than that of the bottom trawl or ROV. During the analysis of the video from this study, we observed that reactions to the SDC by rockfish were minimal, consistent with a previous study with a SDC (Rooper et al., 2010) and a study where a larger towed camera sled was used (Rooper et al., 2007). Fish reactions to underwater vehicles have generally been found to vary with both the species examined (Krieger and Ito, 1999; Lorange and Trenkel, 2006; Ryer et al., 2009) and the type of underwater vehicle used (Stoner et al., 2008). This is an area where more research should be completed in order to gauge the ability of the SDC and other underwater vehicles to accurately measure the height of rockfishes off the seafloor.

Conclusion

Our overall recommendation for verification of target species in acoustic surveys in areas of patchy untrawlable habitat is that a combination of technically advanced stereo-optic equipment and more rugged bottom trawls be used where species identification is likely to be difficult or where many species are found in the water

column. In cases where the rockfish assemblage is dominated by one or two easily distinguishable species, the stereo-optic methods will be the least destructive way to obtain the basic information needed to conduct fisheries acoustic surveys. An important problem highlighted by this research is that species exclusively found in the acoustic dead zone (for example, yelloweye rockfish in this study) will not be able to be assessed acoustically. For these species, alternative methods such as bottom trawls, long-lines, or optical methods using line transect or area swept survey methods will be the only adequate means for estimating the abundance of these fish. Therefore, our results suggest that the selection of appropriate methods for target verification depends on the specific objectives, habitat types, and species complexes being assessed.

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Abstract—Rockfishes (*Sebastes* spp.) are an important component of North Pacific marine ecosystems and commercial fisheries. Because the rocky, high-relief substrate that rockfishes often inhabit is inaccessible to standard survey trawls, population abundance assessments for many rockfish species are difficult. As part of a large study to classify substrate and compare complementary sampling tools, we investigated the feasibility of using an acoustic survey in conjunction with a lowered stereo-video camera, a remotely operated vehicle, and a modified bottom trawl to estimate rockfish biomass in untrawlable habitat. The Snakehead Bank south of Kodiak Island, Alaska, was surveyed repeatedly over 4 days and nights. Dusky rockfish (*S. variabilis*), northern rockfish (*S. polyspinis*), and harlequin rockfish (*S. variegatus*) were the most abundant species observed on the bank. Backscatter attributed to rockfish were collected primarily near the seafloor at a mean height off the bottom of 1.5 m. Total rockfish backscatter and the height of backscatter off the bottom did not differ among survey passes or between night and day. Biomass estimates for the 41 square nautical-mile area surveyed on this small, predominantly untrawlable bank were 2350 metric tons (t) of dusky rockfish, 331 t of northern rockfish, and 137 t of harlequin rockfish. These biomass estimates are 5–60 times the density estimated for these rockfish species by a regularly conducted bottom trawl survey covering the bank and the surrounding shelf. This finding shows that bottom trawl surveys can underestimate the abundance of rockfishes in untrawlable areas and, therefore, may underestimate overall population abundance for these species.

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Evaluation of rockfish abundance in untrawlable habitat: combining acoustic and complementary sampling tools

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Many fish species associate with and find refuge in high-relief substrate, where bottom trawl surveys are ineffective (O'Connell and Carlile, 1993; Yoklavich et al., 2000; Zimmermann, 2003). The bottom trawl survey of the Gulf of Alaska (GOA) conducted by researchers with the NOAA Alaska Fisheries Science Center (AFSC) (von Szalay et al., 2010) routinely encounters areas that are untrawlable because of rough substrate or known hazards to fishing gear on the seafloor. When untrawlable substrate is located at a designated sampling station, an alternate location with suitable substrate is sought nearby (von Szalay et al., 2010). Mean estimates of species abundance from sampling stations are then extrapolated over the entire management area, including known untrawlable areas. Yet rockfish abundance between trawlable and untrawlable areas can vary considerably (Stein et al., 1992; Jagielo et al., 2003; Rooper et al., 2007) and is often lower in trawlable areas than in untrawlable areas (O'Connell and Carlile, 1993; Rooper et al., 2010).

Therefore, extrapolated estimates can be inaccurate.

In habitats that cannot be sampled adequately with trawls, acoustic methods combined with complementary sampling tools may improve rockfish stock assessments by providing more complete and accurate estimates of rockfish populations. Acoustic surveys can cover large areas and much of the water column in a relatively short time, but accurate abundance estimates require consideration of the target species, their diel movements and association with the seafloor, and the type and structure of the substrate. It has been demonstrated that acoustic surveys can be successfully used to assess pelagic rockfish populations in areas of relatively low relief (Wilkins, 1986; Richards et al., 1991; Stanley et al., 2000; Krieger et al., 2001). Cooke et al. (2003) described methods for acoustically sampling fishes in areas of high relief by performing multiple passes at various angles to thoroughly map the seafloor. However, when fish are on or near the bottom in the acoustic dead

zone, (i.e., the near-bottom zone where the echo from the seafloor masks acoustic signals from organisms near the seafloor), a large portion of the population may go undetected (Ona and Mitson, 1996), particularly in areas where the bottom terrain is rough or variable.

Besides the problem of resolving fish backscatter within the dead zone, scientists also must consider the problem of determining the species composition and size distribution of fishes that are detected in that zone. Starr et al. (1996) used a submersible in association with acoustics to estimate rockfish distribution and abundance. Krieger (1992), and Krieger and Ito (1998) used visual surveys from manned submersibles to assess rockfish abundance in untrawlable areas and compared their numbers with those from trawl catches. For surveying in large areas, however, manned submersibles are costly, labor-intensive, and inefficient. Williams et al. (2010) demonstrated the feasibility of using stereo-video drop (i.e., lowered) camera systems for assessing rockfish species and size in untrawlable areas. Ressler et al. (2009) and Rooper et al. (2010) successfully used underwater cameras and echo sounding systems to assess rockfish populations in rocky habitat. However, the species of interest in these studies were far enough above the bottom that assessment in the acoustic dead zone was not necessary.

Rockfishes, of the genus *Sebastes*, constitute a large and diverse assemblage within North Pacific marine ecosystems and are important components of this region's commercial fisheries. Of the rockfish species in the GOA, Pacific ocean perch (*Sebastes alutus*), northern rockfish (*S. polyspinis*), and dusky rockfish (*S. variabilis*) are among the most abundant. They are the only rockfish species supporting commercial fisheries (aside from occasional directed fisheries for the demersal shelf rockfish complex in specific areas), and all 3 species have experienced local depletions within the last decade (Hanselman et al., 2007). In our study on Snakehead Bank, dusky, northern, and harlequin (*S. variegatus*) rockfishes were the most abundant species observed during our surveys and the species on which our analyses focused. Determination of precise population estimates for dusky, northern, and harlequin rockfishes is challenging because these species aggregate in rocky, high-relief areas where it is difficult to conduct trawl surveys to estimate abundance.

Dusky rockfish are managed as part of the pelagic shelf rockfish assemblage and are routinely caught by trawlers on the outer continental shelf at depths of 100–150 m. Dusky rockfish also have been observed on banks and near gullies with hard, rocky habitats containing sponges and corals. Commercial catches of dusky rockfish are primarily located on banks near Yakutat in southeast Alaska and to the east and south of Kodiak Island, Alaska (Lunsford et al., 2009).

Northern rockfish are presently managed as a single stock in the GOA (Heifetz et al., 2009). The preferred habitat of adult northern rockfish in the GOA appears to be hard, rocky, or uneven substrate on relatively shallow rises and banks on the outer continental shelf

at depths of ~75–150 m. One such rise south of Kodiak Island known as Snakehead Bank accounted for 46% of the northern rockfish catch during the 1990s (Clausen and Heifetz, 2002). Northern rockfish stocks on Snakehead Bank have been depleted, and the commercial fishery is nearly absent compared to past effort in this area (Heifetz et al., 2009).

The primary objective of this work, which formed part of a larger study, was to use acoustic and complementary sampling tools to evaluate the feasibility of improving abundance estimates of rockfish species in an untrawlable habitat in the GOA. Other aspects of the larger study, comparing sampling technologies (Rooper et al., 2012 [this issue]) and investigating the use of acoustics for substrate classification (Weber¹), are reported elsewhere in this issue of *Fishery Bulletin* or otherwise available. We used a combination of acoustic backscatter measurements, video observations from a stereo-video drop camera (SDC), a remotely operated vehicle (ROV), and catch composition data from a modified bottom trawl to estimate abundances of rockfish species on Snakehead Bank. To establish whether or not rockfishes are disproportionately abundant in untrawlable areas, we compared estimates of rockfish biomass for the dominant species on Snakehead Bank with those obtained from the AFSC biennial bottom trawl survey.

Materials and methods

This study was conducted during the period of 3–12 October 2009 with 2 vessels at a relatively shallow bank, known locally as Snakehead Bank, located at the GOA shelf break about 74.1 km (40 nautical miles [nmi]) south of Kodiak Island (Fig. 1). The acoustic surveys and ROV deployments were conducted aboard the NOAA Ship *Oscar Dyson*. The SDC and bottom trawl were deployed from the FV *Epic Explorer*. This site was selected because of high historical catches of northern rockfish in the commercial fishery and AFSC bottom trawl survey (Clausen and Heifetz, 2002) and an abundance of rough substrate designated as untrawlable by the AFSC GOA bottom trawl survey (Martin²).

The Snakehead Bank survey initially consisted of 14 parallel transects 9.3 km (5 nmi) long and spaced 2.2 km (1.2 nmi) apart (Fig. 1). Several transects were extended where significant backscatter continued beyond the original endpoints used during the first pass. A pass, defined as a complete survey of all transect lines, was attempted twice—once during daylight hours and again at night—on 4 consecutive days. The number and length of transects surveyed were similar within each pair of passes (day and night) but varied between pairs because deteriorating weather conditions made it impos-

¹ Weber, T. 2011. Unpubl. data. Center for Coastal and Ocean Mapping, Univ. New Hampshire, Durham, NH 03824.

² Martin, M. 2009. Personal commun. Alaska Fisheries Science Center, Seattle, WA 98115.

sible to cover all transects on each successive pass. The core area, or the common area covered on all passes (Fig. 1), was used in further analyses to ensure that similar areas were used in comparisons between passes made on different days and between pairs of passes.

Acoustic equipment and backscatter processing

Acoustic measurements were collected with a calibrated Simrad³ (Kongsberg AS, Horten, Norway) EK60 scientific echo sounding system (Simrad, 2004) with 5 split-beam transducers (18, 38, 70, 120, and 200 kHz) and a Simrad ME70 multibeam echo sounder (Trenkel et al., 2008). The split-beam transducers were mounted on the bottom of a retractable centerboard, positioning the transducers 9.15 m below the water surface during

survey activities. A pulse length of 0.512 ms and ping rate of 1.0 s were used for all EK60 data collections. Nominal half-power beam widths were 7° for the 38-, 70-, 120-, and 200-kHz transducers and 11° for the 18-kHz transducer. Acoustic instruments on the *Oscar Dyson*, other than the split-beam and multibeam systems, were turned off (e.g., the navigational fathometer, Doppler speed log) during acoustic data collections. Data processing and analyses of the acoustic data were performed with Echoview software, vers. 4.70.48 (Myriax Software, Hobart, Tasmania, Australia). The 38-kHz echo sounder was the primary source for the quantitative rockfish backscatter measurements presented here. To measure performance of the EK60 system, acoustic system calibrations with a standard target were conducted by following the methods of Foote et al. (1987).

The echo sounders estimated the distance to the bottom with the amplitude-based algorithm (with a threshold of -36 dB re 1 m^{-1}) implemented in the echo sounder software (Simrad EK60, vers. 2.1.2). The mean of the sounder-detected bottom from all 5 frequencies of the EK60 echo sounder was used as the bottom discrimination line in further data processing (Jones et al., 2011). Acoustic measurements were integrated from 16 m below the surface to the bottom discrimination line. All echograms were examined for bottom integrations. Acoustic backscatter was averaged at 2 resolutions: 185 m (0.1 nmi) horizontal by 1) 0.5 m vertical down to 0.5 m above the bottom discrimination line and 2) 0.25 m vertical from 0.5 m to the bottom discrimination line. All data were exported using an S_V integration threshold of -70 dB re 1 m^{-1} .

Based on calculations from Ona and Mitson (1996), the near-bottom acoustic dead zone calculated with the current system configuration was about 0.3 m at a depth of 100 m. With an additional zone of partial integration (where part of the sampled volume is in the dead zone) equivalent to ~ 0.2 m and a backstep of 0.25 m (to ensure that backscatter from the seafloor is excluded), the total integrator dead zone at a depth of 100 m was ~ 0.7 m above the sounder-detected bottom.

Backscatter was designated to a category (i.e., rockfishes on the bank, deep rockfishes, bubbles, or zooplankton mix) based on backscatter morphology, location on the bank, depth in the water column, and frequency response. Backscatter attributed to rockfishes was assigned to 2 categories based on location in the water column and whether the rockfishes were located on the shallow bank or deeper adjacent shelf break (i.e., bank flanks). Thus, backscatter in one category, hereafter referred to as *rockfishes*

³ Mention of trade names or commercial companies is for identification purposes only and does not imply endorsement by the National Marine Fisheries Service, NOAA.

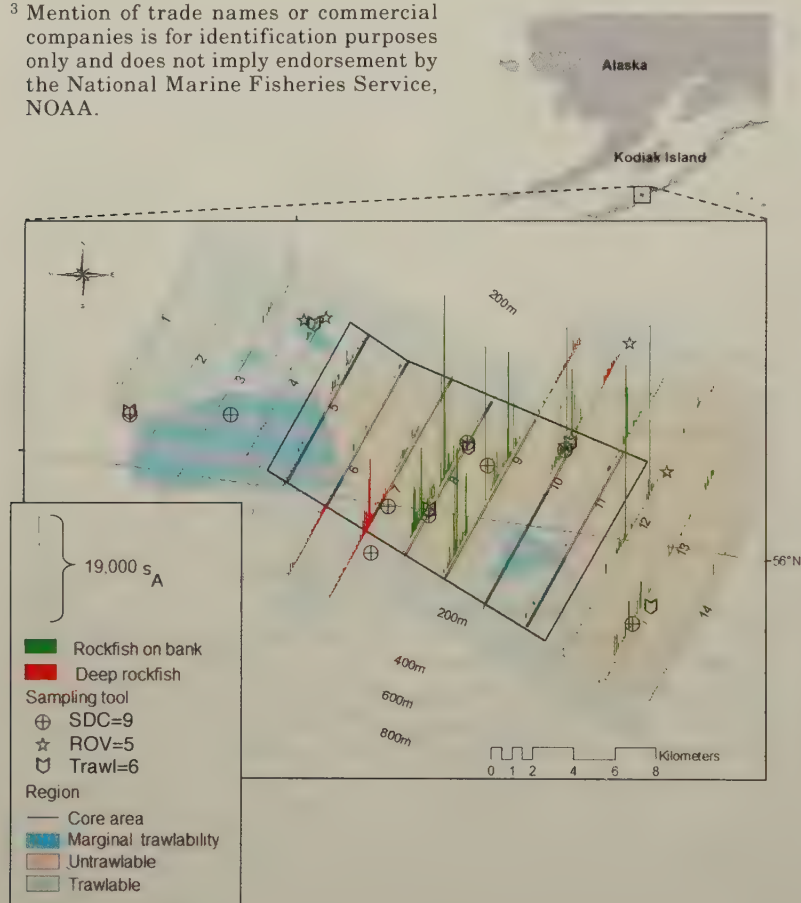


Figure 1

Location of the survey site on Snakehead Bank in the Gulf of Alaska near Kodiak Island, Alaska. Parallel lines represent the full extent of transects surveyed and the core area is represented by the rectangle outlined in black in the middle of this bank. Other colored polygons represent *trawlability*, which was determined with multibeam acoustic backscatter. Symbols indicate sites where the stereo-video drop camera (SDC), remotely operated vehicle (ROV), or bottom trawls were deployed. Green bars depict acoustic backscatter (s_A , $\text{m}^2 \text{nmi}^{-2}$) attributed to rockfishes (species mix) on the bank. Red bars depict rockfishes (e.g., Pacific ocean perch [*Sebastes alutus*]) detected at depths >150 m along the bank flanks. The height of the scale bar for acoustic backscatter represents $19,000 s_A$.

on the bank, indicated rockfishes located on top of the relatively shallow bank (<150 m) within ~5 m of the bottom and represented the dominant species observed by SDC and ROV and captured by trawl. Backscatter in the other category, hereafter referred to as *deep rockfishes*, indicated rockfishes located at depths >150 m and generally >10 m off the bottom over the bank flanks. The deep rockfish backscatter over the bank flanks was attributed to Pacific ocean perch because that was the only species observed in an SDC deployment in that vicinity.

Several areas on the bank contained backscatter that resembled bubble plumes rising from the seafloor. Such backscatter was characterized by comparing the frequency response relative to 38 kHz. The expected volume backscattering strength from rockfishes at 18, 70, 120, and 200 kHz is within 5 dB of the volume backscattering strength at 38 kHz (De Robertis et al., 2010). Any backscatter resembling bubble plumes with a frequency response that differed from the response at 38 kHz by more than 5 dB was classified as bubbles; otherwise, backscatter was classified as rockfishes on the bank or as deep rockfishes over the flanks.

Differences in mean rockfish backscatter for all 8 passes were evaluated with ANOVA. Tests were performed on natural log-transformed data because of unequal variances in the raw data. Differences in rockfish backscatter within pass pairs (between night and day) were evaluated with a paired *T*-test. All tests were considered significant at an alpha level of 0.05.

The mean height above the seafloor of the seafloor, or height off bottom (m), for backscatter attributed to rockfishes on the bank was calculated for each pass with the following formula:

$$\text{Mean height off bottom} = \sum (s_{Ai} \times h_i) / \sum s_{Ai},$$

where s_{Ai} = the nautical area scattering coefficient (MacLennan et al., 2002) in each bin with a resolution of 185×0.5 m (except in the bin closest to the bottom, which was 0.25 m high and offset from the bottom by an additional 0.25 m); and

h_i = the height off bottom of each respective bin.

For rockfish backscatter and height off bottom, each pass was considered a sample unit because data for adjacent transects were not independent. In addition, transects differed in length, and, if transects were used as sample units, the contribution of the shorter transects would be disproportionate compared to the contributions of other transects because shorter transects would receive the same weight as longer ones. Because of these conditions, estimates of sampling variance were expressed as coefficients of variance (CV) with passes as the sample unit, rather than as standard deviations derived from transects as the sample units, and, for that reason, we do not show error bars in our figures.

Stereo-video drop camera

The SDC (for a full description, see Williams et al., 2010) was used to identify and count fish species. Paired still images from 2 video cameras were used to estimate fish length and height off bottom. All SDC deployments were conducted in locations where fish aggregations were identified acoustically. The SDC was maintained at a constant height off bottom by using a live video feed to the surface. The paired cameras were oriented at 30° off horizontal (forward and slightly down), allowing the field of view to extend vertically from the seafloor to ~3 m off bottom. The horizontal field of view surveyed by the cameras (*W*) was ~2.4 m. The distance the SDC covered along the seafloor (*L*) was approximated by using the GPS on the *Epic Explorer*. The area swept during each SDC deployment was calculated as *W*×*L* and the catch per unit of effort (CPUE) for each species was calculated as the number of fish observed per area swept.

All fishes observed in a camera deployment were counted and identified to species when possible. Height off bottom was measured from the seafloor to 2.0 m off the bottom and grouped in 0.5-m increments. Height off bottom was estimated from a single camera for 2 deployments because a malfunction of one of the cameras did not allow stereo measurements of fish length or height off bottom. Height off bottom was compared for single- and stereo-camera counts from deployments where both cameras functioned properly.

Remotely operated vehicle

A Phantom DS4 ROV (Deep Ocean Engineering, Inc., San Jose, California) was used to collect data to verify substrate type, identify species, measure length of dominant rockfishes, and determine species–substrate relationships (for a full description, see Rooper et al., 2012). All measurements were made with a pair of parallel lasers 20 cm apart and a third laser that crossed each parallel laser at specified distances from the cameras. Height off bottom for fishes observed in ROV deployments was estimated as either on the bottom, up to 2.0 m off the bottom, or >2.0 m off the bottom. The ROV was not maintained with a constant field of view above the seafloor; therefore, we did not calculate the area swept and a CPUE for this survey tool.

Modified bottom trawl

Trawl deployments were conducted to collect rockfish specimens for species and size composition for comparison with SDC data (for a full description, see Rooper et al., 2012). The trawl was a modified 4-seam Poly-Nor'Eastern bottom trawl similar to those trawls used by the AFSC in the GOA bottom trawl survey (Stauffer, 2004). The major modifications to the net were heavier netting material in the belly of the net, a footrope with tire gear through the center, and continuous roller gear through the sweeps. Estimates of rockfish densities were not calculated from these trawl deployments

because they were conducted in locations where fish aggregations were acoustically detected and therefore the level of catches would be biased high. Because our trawl deployments were not done at random locations, catch estimates from them could not be compared with results from regular bottom trawl surveys designed to provide estimates of rockfish density and biomass.

Abundance estimation

Fish abundance was estimated for the 3 most abundant rockfish species encountered in the core area covered in our study on Snakehead Bank: dusky, northern, and harlequin rockfishes. Abundance estimates above the acoustic dead zone were calculated for each species and depth layer. These estimates were then combined with abundance estimates from the acoustic dead zone, which were calculated by using 2 different methods (described later in this section), to obtain estimates of total species abundance.

Length-frequency distributions and species compositions were derived from SDC, ROV, and trawl deployments. Length-frequency distributions, backscatter measurements, species compositions, and a target strength (TS) regression (described later in this section) were used to estimate the total number of fish in 1-cm length bins, by following Simmonds and MacLennan (2005). Length-weight relationships obtained from catch data for each species, from AFSC bottom trawl surveys conducted in the summer in the GOA, were used to estimate a biomass for each species and depth layer above the acoustic dead zone.

It was not possible to obtain an estimate of rockfish TS during this study, and no published estimates for the primary species encountered are available. Therefore, the regression described for generic physoclist fishes, $TS = 20 \log_{10} L - 67.5$, where L is fork length (cm) (Foote, 1987), was used as an approximation. Stanley et al. (2000) used this TS relationship for widow rockfish (*S. entomelas*) because it was shown to also agree with several studies on deepwater redfish (*S. mentella*). Rooper et al. (2010) also used the same TS regression for a combination of *Sebastes* species in the Bering Sea. Furthermore, Kang and Hwang (2003) examined *ex situ* TS of Korean rockfish (*S. schlegelii*) and obtained a similar relationship of $TS = 20 \log_{10} L - 67.7$.

Biomass in the 0.7-m acoustic dead zone was calculated by 2 methods to account for the binning of the video observations in 0.5-m increments. The first method used the correction proposed by Ona and Mitson (1996). This correction extrapolates backscatter to the dead zone from a designated zone above the dead zone. The resulting backscatter within the dead zone was apportioned to species based on species composition data from the SDC. The second method for calculating abundance in the dead zone used 2 combinations of depth layers and species ratios from SDC counts (i.e., "1.0-m SDC ratio" and "0.5-m SDC ratio", Fig. 2, B and C). This method, where a constant, weight-specific TS across species and size classes is assumed, used the ratio of species relative abundance from SDC counts in adjacent depth layers to extrapolate abundance from a depth layer above the dead zone to a layer within the dead zone with the following equation:

$$A_{z,j} = (C_{z,j} / C_{z+1,j}) \times A_{z+1,j},$$

where $A_{z,j}$ = the abundance in metric tons of species j in depth layer z ;

$C_{z,j}$ = the relative abundance of species j in depth layer z (from the camera data);

$C_{z+1,j}$ = the relative abundance of species j in the depth layer $z+1$ (also from the camera data); and

$A_{z+1,j}$ = the abundance in metric tons (derived from acoustic measurements) of species j in depth layer $z+1$.

For the "1.0-m SDC ratio," z represents the depth layer of 0–1.0 m and $z+1$ represents the depth layer of 1.0–2.0 m. For the "0.5-m SDC ratio," z represents the depth layer of 0–0.5 m and $z+1$ represents the depth layer of 0.5–1.0 m. Abundance estimates for all depth layers were combined for total biomass values by species and method.

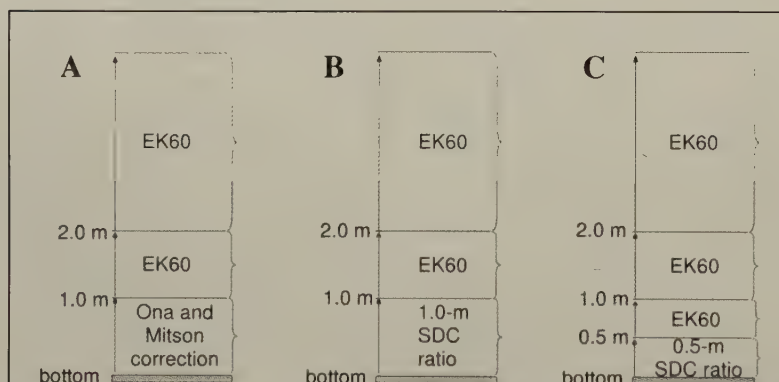


Figure 2

Diagram of depth layers used in calculations of abundances for the dead zone in 2 methods: (A) Ona and Mitson (1996) dead zone correction, as well as (B) extrapolation of abundance from the depth layer of 1.0–2.0 m to the depth layer of 0–1.0 m using the ratio of fishes observed in counts from images collected with the stereo-video drop camera (SDC) and (C) extrapolation of abundance from the depth layer of 0.5–1.0 m to the depth layer of 0–0.5 m by using the ratio of fishes observed in SDC counts. EK60 refers to abundance estimation by using backscatter collected with a Simrad EK60 scientific echo sounder. A dead zone is a near-bottom area where the echo from the seafloor masks acoustic signals from organisms near the seafloor.

Trawlability index

Multibeam acoustic data collected with an ME70 echo sounder were processed to characterize parameters that could potentially be used as an index for *rawlability* (Weber¹). SDC and ROV images were used to verify substrate typing from these multibeam data. The trawlability index was mapped along with EK60 backscatter by using ARCMAP software, vers. 9.3.1 (ESRI, Redlands, California) to determine the amount of area designated to each substrate type and the association between substrate type and fish backscatter.

Results

The core area of the acoustic survey comprised parts of 7 transects (numbers 5–11, Fig. 1) totaling ~59 km (32 nmi) for each of 8 passes. Bottom depths ranged from 63 to 233 m (mean 118 m) over all transects and 63 to 171 m (mean 101 m) within the core area.

Backscatter designation and height off bottom

Most (63%) of the backscatter attributed to rockfishes on the bank was observed within the core area, primarily along the 3 eastern transects (numbers 8–10, Fig. 1). The variation in rockfish backscatter among passes was relatively low ($CV=0.27$, $N=8$, Fig. 3A), and no significant difference was observed in mean rockfish backscatter between day and night passes ($P=0.29$, Fig. 3A).

Counts of fishes off bottom, determined from deployments with only one functional camera, were verified by comparing them with counts from the stereo-video camera deployments where both cameras functioned properly. With the single-camera deployments, ~10% of dusky rockfish and 25% of northern rockfish were closer to the bottom than those same species observed with the stereo-cameras during the same deployments. No harlequin rockfish were seen during the deployments from which single- and stereo-camera comparisons were made. When the deployments during which images were collected from only one camera were not included in analysis, overall abundance estimates decreased ~40% for dusky rockfish and increased 350% for northern rockfish. These differences in abundance estimates resulted from a change in the relative species abundance: 83% of all dusky rockfish and 79% of all harlequin rockfish encountered on all SDC deployments were observed during the 2 single-camera deployments. Because of the relatively minor change in assignments of height off bottom and the large change in species composition and abundance that would result if these data were not included, estimates of height off bottom from single-camera deployments were included in our analyses.

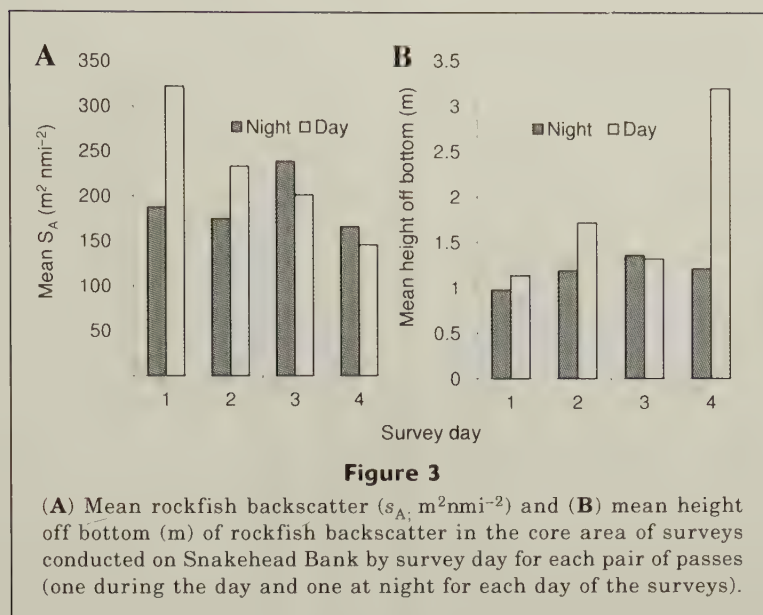


Figure 3

(A) Mean rockfish backscatter (s_A , $m^2 nmi^{-2}$) and (B) mean height off bottom (m) of rockfish backscatter in the core area of surveys conducted on Snakehead Bank by survey day for each pair of passes (one during the day and one at night for each day of the surveys).

The mean height off bottom for backscatter attributed to rockfishes over all passes in the core area was 1.5 m (Fig. 3B). Height off bottom for rockfish backscatter was variable among passes ($CV=0.47$, $N=8$) largely because the height off bottom of backscatter was greater on the last daytime pass of the survey than on other passes.

Species composition

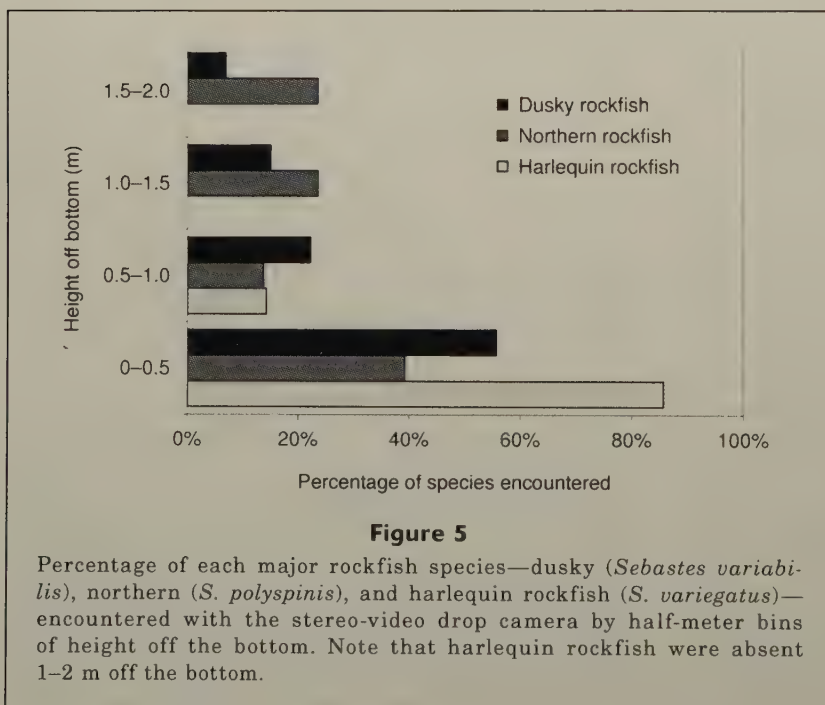
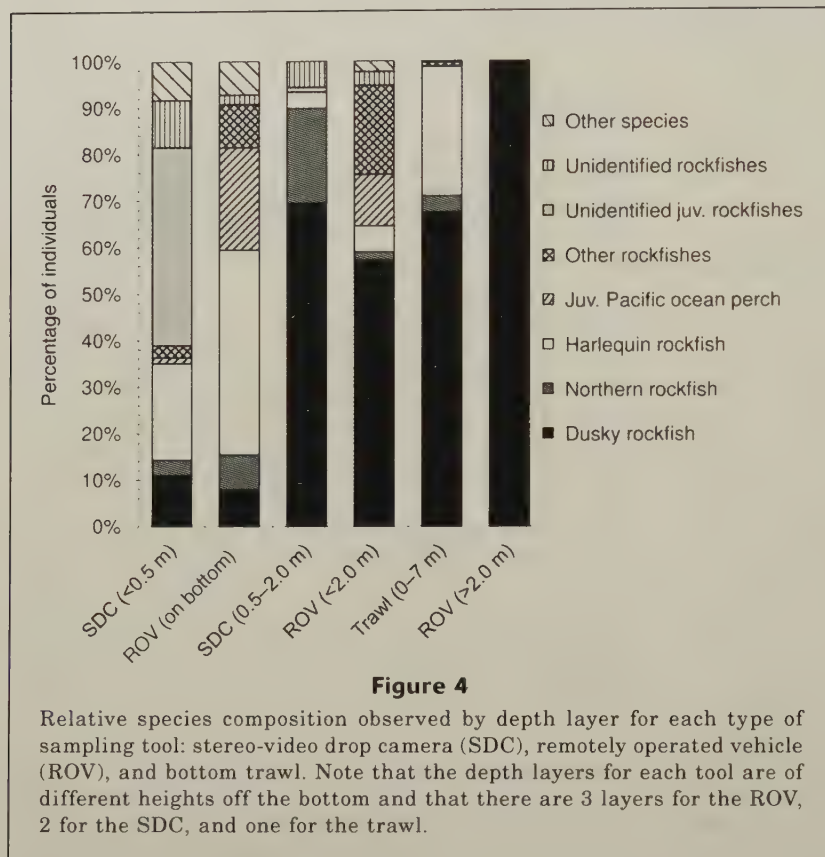
Relatively similar species compositions of the major rockfish species (dusky, northern, and harlequin rockfishes) were observed with the different sampling tools. For all sampling tools, the dusky rockfish was more abundant (40% of individuals for SDC, 51% for ROV, and 67% for trawl) than all other species, and the harlequin rockfish was the second-most observed species (12% of individuals for both SDC and ROV, and 28% for trawl). Aside from juvenile Pacific ocean perch observed with the ROV (12%), northern rockfish was the third-most abundant species (3% of individuals for ROV, 4% for trawl, and 12% for SDC). The SDC observed the highest number of unidentified juvenile (22%) and adult (8%) rockfishes, and the ROV observed the largest number (10) of species identified (for full details, see Rooper et al., 2012).

Stereo-video drop camera

In total, 9 deployments of the SDC were conducted (Fig. 1). More than 3 times as often as any other species, dusky rockfish were observed at heights >0.5 m off the bottom with the SDC (Fig. 4). Although dusky rockfish composed only ~10% of all fishes identified at heights <0.5 m off the bottom (Fig. 4), 56% of all observed dusky rockfish were seen in this depth layer (Fig. 5). Surveyed with the SDC, unidentified juvenile rockfishes composed the largest group (43%) that was observed at heights <0.5 m off the bottom (Fig. 4).

Northern rockfish made up 20% of all fishes encountered at heights >0.5 m off the bottom with the SDC but were a small percentage (3%) of all fishes observed

<0.5 m off the bottom (Fig. 4). However, the majority of northern rockfish (39%) were encountered <0.5 m off the bottom (Fig. 5).



Harlequin rockfish composed 21% of the fishes observed at heights <0.5 m off the bottom with the SDC (Fig. 4). Harlequin rockfish were observed only <1.0 m off the bottom with the SDC and were most prevalent (86%) <0.5 m off the bottom (Fig. 5).

Remotely operated vehicle

The ROV was deployed at 5 sites during the survey (Fig. 1). Of the dusky rockfish observed with the ROV, ~3% were found on the bottom and 11% were seen >2.0 m off the bottom. In contrast, ~65% of harlequin rockfish and ~50% of the northern rockfish observed with the ROV were on the bottom.

Bottom trawl

Trawl deployments were conducted at 6 locations (Fig. 1). More than 98% of the individuals caught in the bottom trawl were from the 3 major rockfish species (dusky, northern, and harlequin rockfishes).

Trawlability and abundance estimates

The trawlability index derived from the multibeam sonar (Weber¹) suggested that the majority (73%) of the core area covered in our survey consisted of untrawlable habitat. Additionally, the majority of the rockfish backscatter (95%) from the core area was located in that untrawlable habitat (Fig. 1).

Only dusky rockfish were found >2.0 m off the bottom on the bank; therefore all rockfish backscatter >2.0 m off the bottom was attributed to that species. The resulting biomass estimate for dusky rockfish observed within the core area >2.0 m off the bottom was 262 metric tons (t).

Both dusky and northern rockfishes were observed 1.0–2.0 m off the bottom; therefore, backscatter in that depth layer was split between these species based on their relative abundance in SDC counts (56% and 33%, respectively). The resulting biomass was 331 t for dusky rockfish and 103 t northern rockfish in the depth layer of 1.0–2.0 m off the bottom.

The majority of all fish species were <0.5 m off the bottom according to SDC counts and ROV observations (Fig. 4). Although the 3 major species considered here composed <40% of all fishes observed <0.5 m off the bottom (Fig. 4), the majority of the observed individuals from these 3 species were encountered in this depth layer (Fig. 5).

The abundance estimates determined by using the Ona and Mitson (1996) dead zone correction for fishes observed <1.0 m off the bottom resulted in an additional 2082 t of dusky rockfish (43% of all fishes in that depth layer) for a total water-column biomass of 2676 t. Harlequin rockfish (13% of all fishes in that depth layer) were the second-most abundant species <1.0 m off the bottom, but their biomass amounted to only 79 t because of their small size. Biomass of northern rockfish <1.0 m off the bottom (8% of all fishes in that depth layer) was 217 t, based on the Ona and Mitson correction method, and total water-column biomass was 321 t (Fig. 6).

The abundance estimate for rockfishes <1.0 m off the bottom determined using the approach of the 1.0-m SDC ratio resulted in an additional 1171 t of dusky rockfish (3.5 times the estimate for the 1.0–2.0-m depth layer) and 117 t of northern rockfish (1.1 times the estimate for the 1.0–2.0-m depth layer). Combining all depth layers resulted in total water-column estimates of 1765 t of dusky rockfish and 220 t of northern rockfish (Fig. 6). Because no harlequin rockfish were observed >1.0 m off the bottom, it was not possible to estimate their biomass with this method.

Abundance estimates determined with the 0.5-m SDC ratio and camera counts in the 0.5–1.0 m depth layer resulted in 574 t of dusky rockfish (70% of all fishes in that depth layer), 90 t of northern rockfish (12% of all fishes in that layer), and 28 t of harlequin rockfish (11% of all fishes in that depth layer). For rockfishes encountered <0.5 m off the bottom, the following estimates were calculated: an additional 1441 t of dusky rockfish (2.5 times the estimate for the 0.5–1.0-m depth layer); 258 t of northern rockfish (2.9 times the estimate for the 0.5–1.0-m depth layer estimate); and 167 t of harlequin rockfish (6 times the estimate for the 0.5–1.0-m depth layer). Summing over all depth layers resulted in total water-column estimates of 2609 t for dusky rockfish, 452 t for northern rockfish, and 195 t for harlequin rockfish (Fig. 6).

The total abundance estimates that resulted from these 3 approaches were within 34% of one another for dusky rockfish, 30% for northern rockfish, and 40% for harlequin rockfish (Fig. 6). Because no specific approach to estimate biomass was clearly superior, the estimates were averaged, and an overall biomass for each species was calculated. The resulting mean bio-

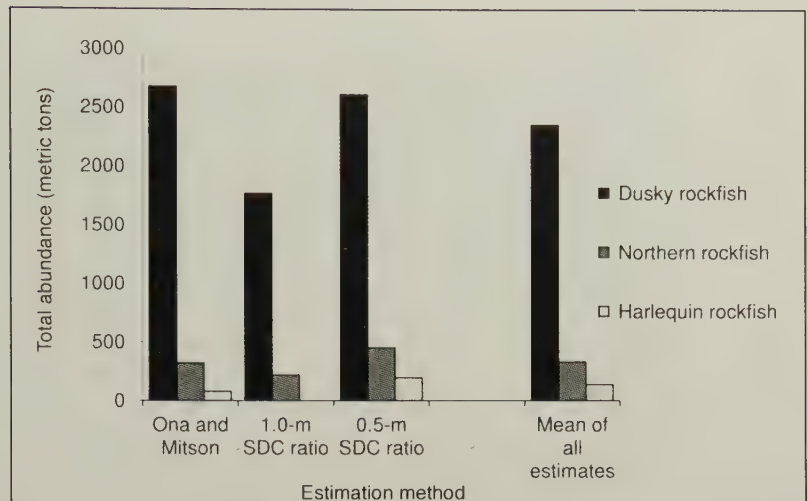


Figure 6

Total abundance values (measured in metric tons of observed fish) for dusky (*Sebastes variabilis*), northern (*S. polyspinis*), and harlequin (*S. variegatus*) rockfishes in the core area of the surveys conducted on Snakehead Bank and calculated with the Ona and Mitson (1996) dead zone correction, 1.0-m and 0.5-m SDC ratios, and with the mean of all of these abundance estimation methods combined.

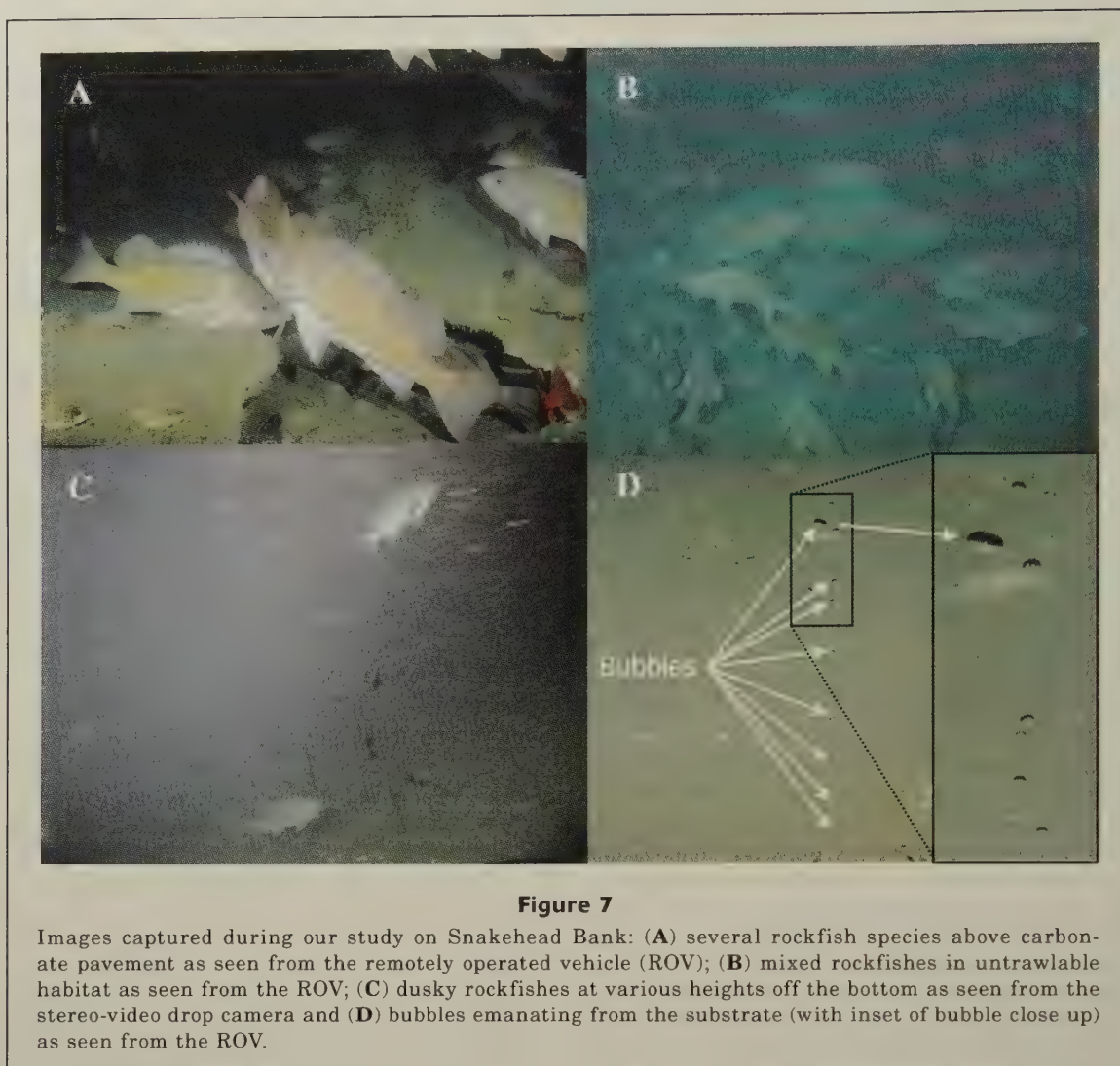
mass estimates were 2350 t for dusky rockfish, 331 t for northern rockfish, and 137 t for harlequin rockfish (Fig. 6).

Backscatter attributed to bubbles

Backscatter at numerous sites within our Snakehead Bank study area resembled rising bubble plumes. These backscatter patterns were visible at all 5 EK60 frequencies and often extended from the seafloor vertically through the lower half of the water column. An ROV deployment in the vicinity of these backscatter verified the presence of bubbles seeping from the seafloor (Fig. 7). Most of the backscatter attributed to bubbles (62%) in the entire survey was recorded within untrawlable areas. At the base of several areas of bubble backscatter, we observed aggregations that appeared to be fish based on echo morphology and frequency response. It was difficult to classify backscatter as either bubbles or fishes. However, the total amount of backscatter attributed to bubbles was <7% of the backscatter attributed to rockfishes. Additionally, most of the backscatter attributed to rockfishes (pass average: 78%) occurred in areas without bubble plumes.

Rock formations, presumably calcium carbonate pavements, and bubbles emanating from the substrate often co-occurred. Subsequent water collections near the bubble seeps verified methane levels in the water column up to 40 times those of atmospheric equilibrium conditions at ambient temperature and salinity (Lilley⁴).

⁴ Lilley, M. 2010. Unpubl. data. School of Oceanography, Univ. Washington, Seattle, WA 98195.



Underwater observations by ROV of areas with hard substrate and bubble plumes in the northwest corner of our study region confirmed that rockfishes were also present in these areas. Underwater video also showed numerous species of rockfishes taking refuge in rocky crevices and under carbonate ledges (Fig. 7).

Discussion

Several sampling tools were used during an acoustic survey to assess the species and abundance of rockfishes on a predominantly untrawlable bank in the GOA. Each tool has advantages and limitations, but, when used together, they can give a more complete picture of habitat and species abundances. Acoustic surveys are excellent means for enumerating midwater organisms of known target strength. However, species that are strongly bottom-oriented are difficult to assess with sonar because of the acoustic dead zone. In addition, this problem is exacerbated in high-relief or sloped terrain

where rockfishes are abundant because the upper extent of the dead zone is determined by where the acoustic beam first encounters the seafloor within the beam footprint (i.e., the shallowest point within the beam). Furthermore, acoustic sampling alone is often insufficient to differentiate between species if multiple species are aggregated or have similar frequency-response or backscattering characteristics. In areas of rough terrain, or for species that are bottom-oriented or aggregated densely, video images can provide a better mechanism to quantify relative species abundance.

Differences observed in the amounts of the rockfish species between the other 3 sampling tools (SDC, ROV, and modified bottom trawl) could be partly explained by the deployment procedures for the different tools. The SDC was lowered to the seafloor and drifted along transects at a consistent height off bottom without altering the camera angle. Because we surveyed in this manner, the SDC sampling effort remained constant for the different depth layers and was viewable up to about 2.0 m off the bottom. The ROV, because it was

powered, did not drift with the *Oscar Dyson* and was capable of observing specific objects of interest and identifying more fishes to species. The added control capability allowed greater flexibility to observe and identify fishes or features of particular interest, such as bubble plumes. Indeed, if not for the precise control of the ROV, the presence of bubble seeps would not have been confirmed. However, when specific objects are investigated, standardization of the viewing field and effort becomes more difficult. Bottom trawl surveys allow for complete species identification and length measurement of captured individuals but do not facilitate allocating catch to specific depth layers. Other aspects of the sampling procedures, such as time to deploy equipment and process samples, difficulty of operation, and cost, have been considered by Rooper et al. (2012).

Generally, northern and harlequin rockfishes observed with the SDC and ROV were smaller than the rockfishes of those species observed with the bottom trawl, indicating size selectivity in the bottom trawl surveys (Rooper et al., 2012). Although the mesh size of the net may allow escape of juveniles and smaller adults, some of the difference in the estimated size distributions between the video and trawl equipment also could be a result of different reactions to the gear by juveniles compared to reactions by adults. Darting into cracks and crevices, juvenile rockfishes appeared to react differently to the ROV (and to a lesser degree to the SDC) than did adults. In contrast, most near-bottom adult rockfishes on the bank did not appear startled or exhibit obvious avoidance behavior to the ROV or SDC, although there was the potential for avoidance or attraction of adult fish to the ROV or SDC outside a camera's field of view (Stoner et al., 2008). If this hiding behavior of juveniles also occurs in response to an approaching trawl and adults show less avoidance behavior, a disproportionate capture of larger fishes may occur.

Despite a locally patchy distribution, abundance in the core area of our survey did not change significantly between passes, indicating that fishes were relatively stable in their geographic distribution over the limited duration of this study. Although most of the backscatter was located in habitat designated as untrawlable, dusky and northern rockfishes also were observed in trawlable areas. Juvenile rockfishes were much more prevalent in untrawlable areas than in trawlable areas, and the harlequin rockfish, which is smaller than the dusky and northern rockfishes, was not seen at all in the trawlable areas. This finding is likely a result of the shelter requirements of juvenile rockfishes and agrees with the observations of Krieger (1992) on unidentified, small (<25 cm fork length) rockfishes in southeast Alaska.

The AFSC GOA bottom trawl survey is conducted biennially to assess the distribution and abundance of the principal groundfish species (von Szalay et al., 2010). Snakehead Bank lies primarily within the Kodiak International North Pacific Fisheries Commission (INPFC) statistical area. Results from the AFSC bottom trawl survey conducted in 2009 indicate that

94% of dusky rockfish observed in the Kodiak INPFC statistical area were found in the depth stratum of 100–200 m that covers an area of 43,333 km² (12,634 nmi²) (von Szalay et al., 2010). The density estimate for dusky rockfish was 8.8 kg/ha in the depth stratum of 100–200 m from the 2009 bottom trawl survey in the Kodiak INPFC statistical area. Our estimate for dusky rockfish from surveys on Snakehead Bank was 167.1 kg/ha, almost 19 times the value of the estimate from the 2009 bottom trawl survey in the Kodiak INPFC statistical area. The difference between our density estimates for Snakehead Bank and the AFSC estimates from the 2009 bottom trawl survey for the entire statistical area is likely attributable to rockfishes being observed predominantly within untrawlable habitat on Snakehead Bank. About 3% of the substrate in the depth stratum of 100–200 m within the Kodiak INPFC statistical area has been designated as untrawlable by the AFSC for its bottom trawl surveys. It is important to note that the designation of trawlability in the AFSC bottom trawl survey does not necessarily equate to our multibeam trawlability index because, unlike our index, the bottom trawl survey's designation is applied to a grid consisting of cells of predefined size. When the higher densities of dusky rockfish from Snakehead Bank were applied to the untrawlable portion of the Kodiak INPFC statistical area in the depth stratum of 100–200 m, the total abundance within that stratum increased by nearly 60% from 38,000 t to 60,000 t. Similar patterns existed for the other 2 rockfish species. The results from the 2009 bottom trawl survey indicated that the majority of northern (54%) and harlequin (97%) rockfishes were observed in the depth stratum of 100–200 m in the Kodiak INPFC statistical area. The density estimate for northern rockfish on Snakehead Bank was 5 times the estimate from the 2009 bottom trawl survey (depth stratum: 100–200 m), and the estimate for Snakehead Bank harlequin rockfish was nearly 60 times greater.

The high rockfish abundances on Snakehead Bank indicate that a substantial quantity of fishes could be overlooked when trawl catches from trawlable areas are extrapolated to larger areas containing untrawlable habitat. Methods for near-bottom measurement are vital to determine accurate estimates of abundance for bottom-oriented species, but quantification becomes particularly difficult when fishes are in complex habitat inaccessible to both sonar and trawls. For the most accurate population assessments, adjustments must be made that account for the bottom-oriented proportion of the stock residing in these complex habitats.

In our study, 2 methods were applied for estimating near-bottom abundance in complex habitat, one of which is applied in 2 different combinations of depth layers. All of the estimation methods use counts from video images to partition backscatter to species and depth layers. The Ona and Mitson (1996) correction essentially calculates the portion of the water column that lies within the dead zone and extrapolates the amount of backscatter in a specified area above the dead zone into that unknown area. This method assumes similar

densities in both the depth layer above and in the dead zone, along with a flat seafloor over the beam width. Yet, as documented in this study, densities can vary within very small distances from the seafloor, and untrawlable areas are, by definition, not flat.

The other method used for estimating abundance in the dead zone relies more on the relative abundance of each species in the dead zone compared to that in the zone above the area where backscatter can reliably be measured. The backscatter was then extrapolated to the dead zone by using the estimated ratio of species relative abundance. This method is much more reliant on estimation of ratios of species relative abundance and the depth layer used in ratios. In our present example, no estimate could be made for harlequin rockfish when the 1.0-m depth layer was used because this species was not present above 1.0 m for extrapolating data down for the dead zone. However, an estimate for harlequin rockfish was possible when the 0.5-m depth layer was used because this species was present in both the depth layers of 0–0.5 m and 0.5–1.0 m.

Abundance estimates calculated in our study rely heavily on video estimates of relative species counts and height off bottom. For that reason, original project plans were to deploy the SDC, ROV, and trawl in the same location successively to obtain a more accurate comparison of their performance. More frequent deployments also were planned, but additional sampling tool deployments and comparisons were not possible because of weather and logistical difficulties.

Rockfishes were associated with carbonate pavements and encountered in the vicinity of bubble seeps (Fig. 7). It is not clear whether this apparent relationship is a result of the particular substrate type in the vicinity of the plumes or the bubble plumes themselves. To help determine the importance of carbonate pavements as rockfish habitat, more observations are needed to characterize this association and describe the geographical distribution of this habitat type in the GOA.

Conclusions

We examined the complexity of methods for obtaining accurate abundance estimates for species that are bottom-oriented and have an affinity for complex habitat. This study shows that an adequate survey of dominant species in untrawlable terrain can be performed with acoustic instruments in conjunction with an SDC-based sampling system. Expanding such a survey to a geographic area larger than the one used in our study would be reasonable once suitable descriptors for substrate habitat classification are employed to characterize an area and enable untrawlable locations to be specifically targeted. Because of the patchy nature of the schooling behavior of fishes and their attraction to specific topographical features, additional targeted camera effort would improve allocation of species distribution and counts. Combined biomass estimates from the trawlable

and untrawlable areas would then provide a picture of species abundances that is much more representative over the entire management area than current practice estimates in which only data from trawlable areas are used.

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Abstract—Estimating rare events from zero-heavy data (data with many zero values) is a common challenge in fisheries science and ecology. For example, loggerhead sea turtles (*Caretta caretta*) and leatherback sea turtles (*Dermochelys coriacea*) account for less than 1% of total catch in the U.S. Atlantic pelagic longline fishery. Nevertheless, the Southeast Fisheries Science Center (SEFSC) of the National Marine Fisheries Service (NMFS) is charged with assessing the effect of this fishery on these federally protected species. Annual estimates of loggerhead and leatherback bycatch in a fishery can affect fishery management and species conservation decisions. However, current estimates have wide confidence intervals, and their accuracy is unknown. We evaluate 3 estimation methods, each at 2 spatiotemporal scales, in simulations of 5 spatial scenarios representing incidental capture of sea turtles by the U.S. Atlantic pelagic longline fishery. The delta-lognormal method of estimating bycatch for calendar quarter and fishing area strata was the least biased estimation method in the spatial scenarios believed to be most realistic. This result supports the current estimation procedure used by the SEFSC.

Evaluating methods for estimating rare events with zero-heavy data: a simulation model estimating sea turtle bycatch in the pelagic longline fishery

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Fishery scientists and ecologists often must make inferences from data with many zero values and high variance. For example, studies of the detection or capture of protected species or infrequently encountered commercial species result in data sets that contain many zeros and few positive values with a skewed distribution (Martin et al., 2005; Sileshi, 2006). Analyzing such zero-heavy data sets (data sets with many zero values) poses unique challenges that are not always met, perhaps, because method suitability has not been explored fully or because of deference to familiar methods (Walters, 2003; Martin et al., 2005; Sileshi, 2006). It is not uncommon for scientists to use familiar statistical methods even when it may be impossible to meet model assumptions (Walters, 2003; Sileshi, 2006). Additionally, transformations often are employed to overcome violations of the errors' assumed variance-mean relationship, but transformations will not ameliorate the problems associated with zero-heavy data (Martin et al., 2005). Biased estimates and incorrect conclusions can result from not accounting for excess zeros and using models with inappropriate assumptions (Martin et al., 2005).

However, interest is growing in analyzing data with excess zeros and

in estimating rare events because more appropriate analyses can provide more accurate results (Martin et al., 2005). If scientists use the most appropriate analysis method for a system, they are more likely to obtain the best available estimate for making management decisions for their study system. In this article, we evaluate several methods for making inferences from zero-heavy data sets in the context of estimating fleetwide bycatch of sea turtles. By evaluating method performance, we identify the most suitable estimation method in a variety of fishery scenarios.

The U.S. Atlantic pelagic longline fishery targets swordfish (*Xiphias gladius*) and tuna (*Thunnus* spp.) in the Atlantic Ocean, Caribbean Sea, and Gulf of Mexico. From 2005 to 2007, longlines were used to catch approximately 73% of swordfish, 84% of yellowfin tuna (*Thunnus albacares*), and 90% of bigeye tuna (*Thunnus obesus*) domestic landings by weight nationwide, where fishing gear was specified (NMFS¹). However, swordfish and tuna constituted less than

¹ NMFS (National Marine Fisheries Service). 2009. Annual commercial landings by gear type, http://www.st.nmfs.noaa.gov/st1/commercial/landings/gear_landings.html, accessed 12 May.

half of the observed catch from the U.S. Atlantic pelagic longline fishery between 1992 and 2002 (Beerkircher et al., 2004). The rest of the catch was incidental bycatch. Sharks, rays, and finfishes composed the majority of bycatch during this period, and the incidental capture of sea turtles and marine mammals made up about 1% of the observed catch (Beerkircher et al., 2004). For example, out of the 944 observed sets in 2007, 114 caught a sea turtle (Fairfield and Garrison, 2008). A fishing set is a single deployment of fishing gear; a vessel on average fishes 6 sets per 9-day trip (NMFS, 2006).

Although the incidental capture of the loggerhead sea turtle (*Caretta caretta*) and leatherback sea turtle (*Dermochelys coriacea*) is rare, it is notable because these species are protected under the Endangered Species Act (ESA) of 1973: the leatherback sea turtle is listed as endangered and the loggerhead sea turtle is listed as both endangered and threatened. The endangered distinct populations of the loggerhead sea turtle include one in the northeast Atlantic, and the distinct populations listed as threatened include a population in the south Atlantic and another in the northwest Atlantic.

Because the sea turtles that are caught by the U.S. Atlantic pelagic longline fishery are protected under the ESA, scientists at the Southeast Fisheries Science Center (SEFSC) of the National Marine Fisheries Service estimate the number caught annually. These annual bycatch estimates are compared with the fishery's incidental take statement (ITS), which stipulates the maximum number of sea turtles the fishery may catch incidentally before formal consultation under section 7 of the ESA must be undertaken. If the maximum number stipulated in the ITS is exceeded for a turtle species, the SEFSC must assess whether the fishery is jeopardizing the survival of that turtle species and, consequently, how the fishery is allowed to proceed (McCracken, 2004). Therefore, accurate and precise estimates are necessary for both sea turtle conservation and appropriate fishery management.

The SEFSC bases its estimates of sea turtle bycatch on 2 sources of data: logbooks kept by vessel captains and records made by independent observers deployed on ~8% of vessels (Beerkircher et al., 2004). Vessel captains are required to keep logbooks and record information about fishing gear, location, effort, target, and catch. Observers are charged with collecting unbiased data that are representative of the total catch composition (Crowder and Murawski, 1998; Fairfield and Garrison, 2008). To estimate fleetwide sea turtle bycatch, bycatch rates are extrapolated from observer data and on the basis of observer logbook data are applied to unobserved fishing sets (Fairfield and Garrison, 2008). Generally, bycatch is estimated by identifying a relationship between fishing effort or environmental characteristics and the number of turtles caught on observed fishing sets and then by assuming that that relationship holds for unobserved sets.

The estimation methods essentially can be categorized as sample-based estimators or model-based predictors. For sample-based estimators, sampling probabilities

are assumed but, for the most part, assumptions are avoided regarding the structure of the target population and features being estimated. These estimators allow the observed bycatch rate to be raised to fleetwide estimates on the basis of total reported fishing effort. Sample-based estimators are usually less efficient (i.e., they require more samples to achieve a specified level of performance) than model-based predictors, where a statistical model of bycatch is assumed. The statistical model used in model-based predictors represents the process that is generating the response variable as a function of explanatory variables (McCracken, 2004). In our example, parameters can be estimated with the data from observed sets and used to relate the explanatory variable values recorded in the logbooks to the number of sea turtles caught. These relationships can be used to estimate the number of turtles caught on unobserved sets.

Current SEFSC estimates of sea turtle bycatch by the U.S. Atlantic pelagic longline fishery have wide confidence intervals, and their accuracy is unknown. Consequently, it is difficult to determine the level of bycatch in a single year and the trend over time, and insufficient bycatch information impedes management. The ability of the SEFSC to estimate bycatch—and, thus, of the NMFS to manage the fishery and conserve protected species—may be improved if alternative estimation methods are systematically compared and the most suitable estimation method is identified. Evaluation of estimation methods with regards to frequently encountered data complexities, such as small sample size (8% observer coverage), overdispersion (greater variance than expected), excess zeros (many observed sets without bycatch), and hierarchical observations (sampling fishing sets within trips), is particularly warranted.

In this study, we evaluated 2 of the most prevalent methods for estimating rare events with zero-heavy data: the delta-lognormal method, a sample-based estimator (Pennington, 1983); and the generalized linear model (GLM), a model-based predictor (Lindsey, 1997), in the context of sea turtle bycatch in the U.S. Atlantic pelagic longline fishery. The SEFSC has used the delta-lognormal method to estimate sea turtle bycatch in the U.S. Atlantic pelagic longline fishery since 1997, but, in recent years, the SEFSC has considered switching to a GLM approach (Fairfield and Garrison, 2008; Garrison²).

In comparison, the Southwest Fisheries Science Center (SWFSC) and Pacific Islands Fisheries Science Center (PIFSC) have estimated sea turtle bycatch in the U.S. Pacific pelagic longline fishery. The SWFSC used a survey sampling theory in 1994 and 1995 and a regression tree model in 1996 (Skillman and Kleiber, 1998). In 2000, McCracken (2004) of the PIFSC completed the first official report that systematically examined different methods for estimating sea turtle bycatch in the U.S. pelagic longline fishery, although sea turtle

² Garrison, L. P. 2009. Personal commun. National Marine Fisheries Service Southeast Fisheries Science Center, Miami, FL.

bycatch has been estimated since 1992. McCracken (2004) determined that the GLM with a Poisson error distribution and its generalized additive model (GAM) counterpart were the most appropriate methods for estimating sea turtle bycatch in the U.S. Pacific pelagic longline fishery from 1994 to 1999. However, McCracken did not consider the delta-lognormal method, and data from the Atlantic fishery were not analyzed. The Pacific fishery was closed in 2000 and reopened in 2004. Since then, observer coverage has been at least 20%, and bycatch has declined to the point that it is not necessary to model bycatch; instead, the Horvitz-Thompson estimator has been used by the PIFSC (McCracken, 2004).

The goal of this study was to evaluate delta-lognormal and GLM performance under a variety of spatial fishery scenarios to identify the more suitable estimation method. We built a simulation model representing a range of spatial interactions of sea turtles with the U.S. Atlantic pelagic longline fishery and used the delta-lognormal method, a generalized linear model with a Poisson error distribution (GLM-P), and a GLM with a negative binomial error distribution (GLM-NB), each at 2 spatiotemporal scales, to estimate the number of turtles caught. By comparing these estimates to the total number of turtles caught in the simulation, we were able to systematically evaluate the performance of each method.

Materials and methods

To represent sea turtle bycatch by the U.S. Atlantic pelagic longline fishery, we constructed a simulation

model that included 5 spatial scenarios with various distributions of sea turtles and fishing sets (Fig. 1). The simulation model included both SEFSC data and model assumptions based on the current understanding of fishery and sea turtle behavior (Table 1). Observers were simulated on 8% of the fishing sets, and each estimation method was applied to every spatial scenario. The estimation methods were evaluated by comparing the estimated amount of bycatch to the total simulated amount of bycatch. The simulation model was run 1000 times for each of the 5 spatial scenarios, enabling a comprehensive evaluation of the performance of the estimation methods.

The empirical and theoretical foundation of model assumptions

Fishery-independent data on sea turtle spatial distributions are limited to a few satellite-tracked individuals, at most 60 turtles in a study but typically fewer than 20 (Godley et al., 2007), and aerial surveys (Epperly et al., 1995; McClellan, 1996; McDaniel et al., 2000; Goodman et al., 2007). Small sample size, short study durations (typically less than one year), and nonrepresentative sampling of ages and sexes make satellite tracking data unsuitable for our study (Godley et al., 2007). Moreover, inference from aerial surveys can be difficult because of the high percentage of time that turtles spend submerged and variability in turtle surfacing behavior related to season and location (Byles, 1988; Nelson, 1996; Mansfield, 2006; Goodman et al., 2007).

Because fishery-independent data were not suitable for our objectives, we considered fishery-dependent data. These data indicate that sea turtles clump (i.e., tend to concentrate in certain areas rather than occur equally spaced or spaced with uniform probability), especially in productive areas of the ocean. Currents, frontal regions, and some bathymetric features often are associated with enhanced productivity and prey aggregation, and turtles exhibit a clumping pattern in response to these features when they forage (Williams et al., 1996; Witzell, 1999; Gilman et al., 2006). Environmental features, such as major current systems and gradients in temperature, chlorophyll, and salinity, also seem to influence the clumping of turtles, as well as swordfish (Bigelow et al., 1999; Polovina et al., 2000; Lewison et al., 2004). However, turtle distributions appear to vary seasonally and between species. Gardner et al. (2008) found that, for most of the year, loggerhead and leatherback bycatch locations were not completely random

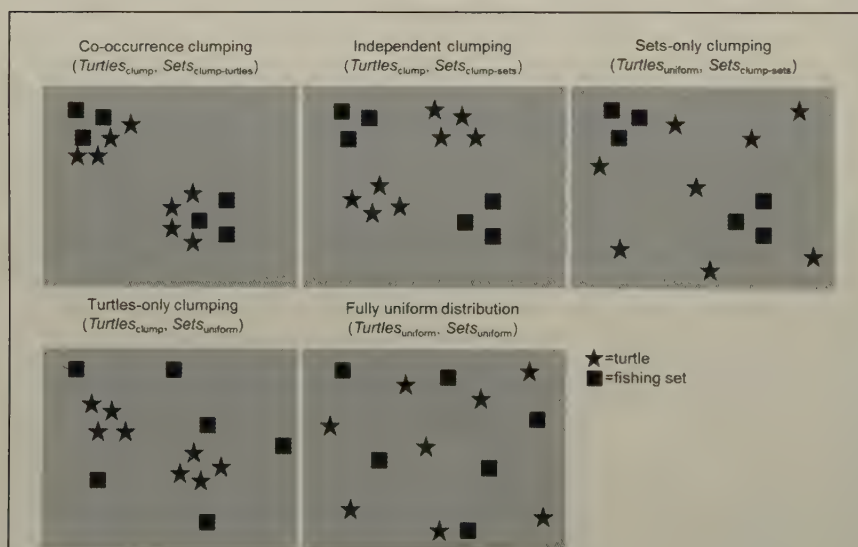


Figure 1

The 5 spatial scenarios depicting interactions between the U.S. Atlantic pelagic longline fishery and sea turtles that were included in the simulation model used in our study. The panels proceed left to right from the scenario considered most realistic at the top left to the least realistic at the bottom middle. ★=turtle. ■=fishing set.

Table 1

Data from the National Marine Fisheries Service (NMFS) Southeast Fisheries Science Center (SEFSC) and the scientific literature that were used in the simulation model built to represent interactions of sea turtles with the U.S. Atlantic pelagic longline fishery.

Model features based on existing data		Values	Source
Mean number of annual fishing sets	8000		SEFSC
Mean mainline length of fishing sets	50 km		SEFSC
Attributes of fishing sets	Set number within a trip, mainline length, target species, presence of light stick, number of hooks, sea-surface temperature, fishing area, date, latitude, longitude		SEFSC
Spatial scenarios of fishing sets	Clumping apparent in location records		SEFSC
Observer coverage	8%		SEFSC
Spatial and temporal variation in fishing effort and bycatch	Variation across 4 calendar quarters and 10 fishing areas in the Atlantic Ocean, Caribbean Sea, and Gulf of Mexico		SEFSC
Probability of sea turtle capture	Variation across calendar quarters and fishing areas		SEFSC
Density of sea turtles	0.5 turtles/km ²		Byles (1988); Nelson (1996); Mansfield (2006); Goodman et al. (2007)
Spatial scenarios of sea turtles	Clumping related to currents, frontal regions, bathymetric features, and prey		Williams et al. (1996); Bigelow et al. (1999); Witzell (1999); Polovina et al. (2000); Lewison et al. (2004); Gilman et al. (2006); Gardner et al. (2008)
Clumping area	90×90 km		Gardner et al. (2008)

and that there seemed to be increased clumping from July to October. Also, clumping was more pronounced with loggerheads than with leatherbacks (Gardner et al., 2008).

Therefore, we modeled clumped and uniformly random sea turtle distributions. Although existing data indicate that turtles clump, very little information about the spatial extent or density of clumps is available. A density estimate of 0.5 turtles/km² was assumed for modeling because it is an intermediate value based on the estimates available in the scientific literature, bearing in mind that an individual turtle may surface and be available to aerial surveys 5.3% to 30% of the time (Byles, 1988; Nelson, 1996; Mansfield, 2006; Goodman et al., 2007).

As for fishing sets, SEFSC maps of longline set locations suggest that sets do not have a uniformly random distribution (Fairfield and Garrison, 2008). For analysis, the SEFSC has divided the Atlantic Ocean, Caribbean Sea, and Gulf of Mexico into 10 geographic regions or statistical areas, and the agency estimates bycatch in each area for each calendar quarter and then sums these estimates to generate a total annual

estimate. Sets appeared clumped whether their distribution was considered across all fishing areas or within a single fishing area. However, the mechanism behind this clumping is not well understood. We modeled 2 possible scenarios: 1) fishing sets clump in the same areas in which sea turtles clump and 2) sets clump independently of turtles. The first scenario could occur if both fishermen and turtles target productive areas of the ocean. The latter could result from either fishermen or turtles imperfectly targeting productive areas or clumping based on another cue. For example, fishermen might aggregate from peer influence.

The spatial scenarios with clumped sets were expected to be most realistic, but considering the amount of uncertainty in the nature of the interactions of sea turtles with the pelagic longline fishery, we thought it useful to analyze other distributions as well. For example, a scenario with uniformly random turtles and sets served as a null model. Further, the results from spatial scenarios considered less realistic for the interactions of sea turtles with the U.S. Atlantic pelagic longline fishery could illuminate general properties

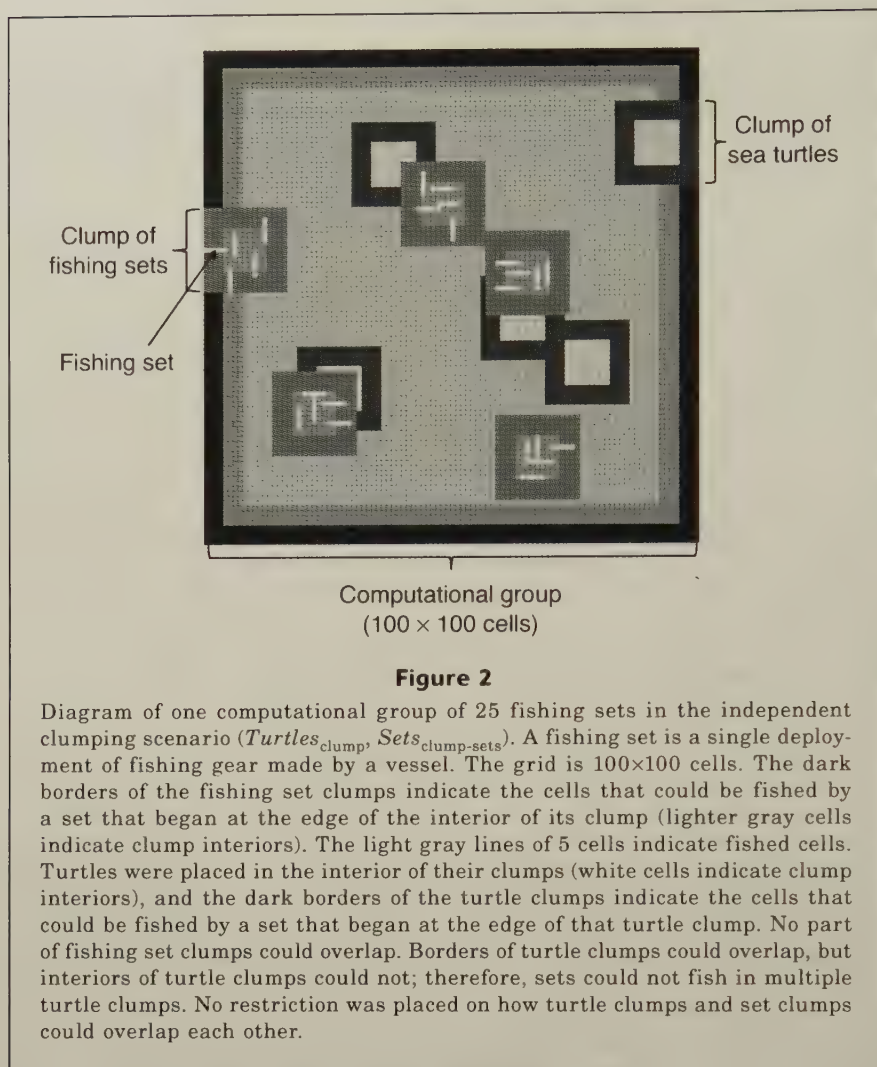


Figure 2

Diagram of one computational group of 25 fishing sets in the independent clumping scenario ($Turtles_{clump}$, $Sets_{clump-sets}$). A fishing set is a single deployment of fishing gear made by a vessel. The grid is 100x100 cells. The dark borders of the fishing set clumps indicate the cells that could be fished by a set that began at the edge of the interior of its clump (lighter gray cells indicate clump interiors). The light gray lines of 5 cells indicate fished cells. Turtles were placed in the interior of their clumps (white cells indicate clump interiors), and the dark borders of the turtle clumps indicate the cells that could be fished by a set that began at the edge of that turtle clump. No part of fishing set clumps could overlap. Borders of turtle clumps could overlap, but interiors of turtle clumps could not; therefore, sets could not fish in multiple turtle clumps. No restriction was placed on how turtle clumps and set clumps could overlap each other.

of the estimation methods that are relevant to other problems with the management of natural resources.

General structure of the simulation model

Much remains unknown regarding the spatial distributions of sea turtles, how fishermen decide where to fish, and the nature of interactions of sea turtles with fishing sets in time and space. Therefore, we designed several spatially explicit scenarios to address the uncertainty and variation in interactions of sea turtles and the fishery. Five spatial scenarios were modeled (Fig. 1): 1) co-occurrence clumping ($Turtles_{clump}$, $Sets_{clump-turtles}$); 2) independent clumping ($Turtles_{clump}$, $Sets_{clump-sets}$); 3) sets-only clumping ($Turtles_{uniform}$, $Sets_{clump-sets}$); 4) turtles-only clumping ($Turtles_{clump}$, $Sets_{uniform}$); and 5) fully uniform distribution ($Turtles_{uniform}$, $Sets_{uniform}$).

Details of model construction

In each simulation, the number of fishing sets that we modeled was 8000, which was approximately the aver-

age number of sets reported annually to the SEFSC from 2005 to 2007 (Walsh and Garrison, 2006; Fairfield-Walsh and Garrison, 2007; Fairfield and Garrison, 2008), the first 3 years after NMFS regulations mandated a change from J-hooks to circle hooks for the longline fishery (Watson et al., 2005). Circle hooks were required to reduce the number of sea turtles caught and the severity of their injuries. However, rather than simulating 8000 sets at once, we divided the 8000 sets into computational groups of 25 sets for convenience (Fig. 2). The computational groups of 25 sets were used to distribute turtles and sets, place observers, and simulate bycatch, but bycatch estimates were not made at this scale.

With the Atlantic Ocean, Caribbean Sea, and Gulf of Mexico divided into 10 geographic regions and bycatch estimates for these statistical areas made by the SEFSC for each calendar quarter, bycatch estimates are made in 40 quarter-area strata. Bycatch rates were expected to vary across these strata; therefore, we also modeled strata (Table 2). For each stratum, we calculated the average number of sets reported to the SEFSC from 2005

Table 2

The number of computational groups of 25 fishing sets and bycatch probabilities per stratum that were included in the simulation model built to represent interactions of sea turtles with the U.S. Atlantic pelagic longline fishery. Bycatch probabilities differed between scenarios with clumping turtles and scenarios with turtles placed with a uniform probability because of the different turtle densities. Clumping means that turtles tend to concentrate in certain areas rather than occur equally spaced or spaced with uniform probability. The SEFSC estimates bycatch of sea turtles for each of 4 calendar quarters (Q1 through Q4) and for each of 10 geographic regions or fishing areas. The SEFSC uses the following names for these fishing areas: CAR=Caribbean, FEC=Florida East Coast, GOM=Gulf of Mexico, MAB=Mid-Atlantic Bight, NCA=North Central Atlantic, NEC=Northeast Coastal, NED=Northeast Distant, SAB=South Atlantic Bight, SAR=Sargasso Sea, and TUN=Tuna North. We simulated 32 quarter-area strata because 8 strata were without fishing or observer coverage from 2005 to 2007.

Simulated quarter-area stratum	SEFSC quarter-area stratum basis	Number of computational groups simulated	Bycatch probability: turtles uniformly random	Bycatch probability: turtles clumping
1	Q1-CAR	3	6.17×10^{-3}	2.50×10^{-4}
2	Q1-FEC	8	1.15×10^{-2}	4.65×10^{-4}
3	Q1-GOM	39	2.36×10^{-3}	9.55×10^{-5}
4	Q1-MAB	6	5.20×10^{-3}	2.11×10^{-4}
5	Q1-NCA	1	2.74×10^{-3}	1.11×10^{-4}
6	Q1-SAB	5	8.98×10^{-3}	3.64×10^{-4}
7	Q1-SAR	4	3.09×10^{-3}	1.25×10^{-4}
8	Q1-TUN	1	2.74×10^{-3}	1.11×10^{-4}
9	Q2-CAR	1	6.17×10^{-3}	2.50×10^{-4}
10	Q2-FEC	7	2.74×10^{-3}	1.11×10^{-4}
11	Q2-GOM	40	5.57×10^{-3}	2.25×10^{-4}
12	Q2-MAB	10	5.20×10^{-3}	2.11×10^{-4}
13	Q2-NCA	1	2.74×10^{-3}	1.11×10^{-4}
14	Q2-NEC	2	1.32×10^{-2}	5.33×10^{-4}
15	Q2-NED	1	2.59×10^{-2}	1.05×10^{-3}
16	Q2-SAB	19	8.98×10^{-3}	3.64×10^{-4}
17	Q2-TUN	2	2.74×10^{-3}	1.11×10^{-4}
18	Q3-FEC	6	7.11×10^{-3}	2.88×10^{-4}
19	Q3-GOM	38	1.07×10^{-3}	4.35×10^{-5}
20	Q3-MAB	24	3.14×10^{-3}	1.27×10^{-4}
21	Q3-NEC	12	1.99×10^{-2}	8.08×10^{-4}
22	Q3-NED	12	2.23×10^{-2}	9.02×10^{-4}
23	Q3-SAB	5	8.98×10^{-3}	3.64×10^{-4}
24	Q3-TUN	2	2.74×10^{-3}	1.11×10^{-4}
25	Q4-FEC	3	7.11×10^{-3}	2.88×10^{-4}
26	Q4-GOM	31	1.02×10^{-2}	4.14×10^{-4}
27	Q4-MAB	23	7.26×10^{-3}	2.94×10^{-4}
28	Q4-NCA	2	6.17×10^{-3}	2.50×10^{-4}
29	Q4-NEC	3	2.96×10^{-2}	1.20×10^{-3}
30	Q4-NED	5	8.98×10^{-3}	3.64×10^{-4}
31	Q4-SAB	2	2.35×10^{-2}	9.52×10^{-4}
32	Q4-SAR	2	2.74×10^{-3}	1.11×10^{-4}

to 2007 and rounded to multiples of 25 to determine the number of computational groups of 25 sets that would be modeled per stratum.

Each computational group of 25 sets was modeled as a grid of 100×100 cells. Sea turtles and fishing sets were assigned coordinates (x, y) depending on the spatial scenario. The details of the procedures are described

in the following sections. Modeled sets covered 5 cells—an initial cell and 4 cells either up, right, down, or left—because the average longline set covers about 50 km (mean 47 km, minimum 32 km, maximum 64 km) (Witzell, 1999; Beerkircher et al., 2004; Gilman et al., 2006). Hence, modeled cells were conceptualized as 10×10 km.

Co-occurrence clumping scenario In spatial scenarios with clumped fishing sets, we modeled computational groups with 5 clumps of 5 sets each. Sea turtles also were aggregated in 5 clumps for clumping scenarios. Each clump was based around a block of 9×9 cells. This use of clumps of 90×90 km was consistent with the results of Gardner et al. (2008), who reported that turtle bycatch distributions were found to span 30–200 km.

We modeled the density of sea turtles as declining with distance from the center of a clump. We selected x and y coordinates for the seed of the first turtle clump with uniform probability. To accentuate clumping, we placed turtles within a clump so that the coordinates closer to the seed had a greater probability: $\text{Prob}(X=X_{\text{seed}})=0.2$, $\text{Prob}(X=X_{\text{seed}+1})=0.16$, $\text{Prob}(X=X_{\text{seed}+2})=0.12$, $\text{Prob}(X=X_{\text{seed}+3})=0.08$, $\text{Prob}(X=X_{\text{seed}+4})=0.04$. Assuming a density of 0.5 turtles/km², we placed an average of 50 turtles/cell or 4050 turtles/clump and 20,250 turtles in the entire grid of 100×100 cells. Subsequent clump seed coordinates were selected so that a set could not fish in multiple turtle clumps.

In the spatial scenario with fishing sets and sea turtles clumped in the same areas, the co-occurrence clumping scenario, the clumps (9×9 cells) for the sets and clumps (9×9 cells) for the turtles were identical. Each fishing set began within the 9×9 cells of its clump and then moved 4 cells up, right, down, or left. A set could leave the 9×9 cells of its clump during fishing. However, clumps were designed with 9×9 cells so that a fishing set that began in a clump's center could move in any direction and remain inside its clump. For each of the 5 fishing sets in a clump, the direction of fishing (up, right, down, or left) was determined by the number of turtles that would be encountered in each direction.

To determine the initial coordinates of fishing sets, we tallied the number of sea turtles in each x coordinate of the clump. This tally was used to construct a probability for set placement by dividing the number of turtles with a particular x coordinate by the total number of turtles. The same was done for the y coordinates. To determine the direction of fishing, we tallied the number of turtles that would be encountered by a set moving right, left, up, or down. These 4 counts were summed, and the number encountered in each direction was divided by the total to obtain a probability of moving in each direction. The more turtles that would be encountered, the greater the probability a set would fish in that direction. This algorithm mimicked a situation where more turtles are in the productive areas that fishermen are targeting than in other areas.

Independent clumping scenario The 2 features that distinguish the independent clumping scenario from the co-occurrence clumping scenario are the following: 1) the clumps (9×9 cells) for fishing sets and turtles were placed independently and 2) the direction of fishing was influenced by the number of sets in each of the 4 directions. That is, there was a positive relationship between the probability a set would fish in a particular

direction and the proximity to other sets in that direction. The smaller the distance to other sets, the greater the probability the set would fish in that direction. This algorithm is consistent with fishermen aggregating because of peer influence.

Initial x and y coordinates were selected for the seed of the first fishing set clump with uniform probability. We also selected x and y coordinates for the starting positions of each of the 5 sets in a clump with uniform probability. Each set had a greater probability of moving in the direction where there were more sets. We first considered $\text{Set}_0\text{Cell}_0$, the first cell in the first set. We calculated the distances from $\text{Set}_0\text{Cell}_0$ to $\text{Set}_i\text{Cell}_0$, where $i = 1$ to 4, and summed these distances. We calculated the distances from $\text{Set}_0\text{Cell}_{1R}$, the cell to the right of the initial fishing cell, to $\text{Set}_i\text{Cell}_0$ and added these distances to the distances from $\text{Set}_0\text{Cell}_0$ to $\text{Set}_i\text{Cell}_0$. We continued to calculate the distances to $\text{Set}_i\text{Cell}_0$ if Set_0 fished to the right and summed the distances. This algorithm gave the distance from Set_0 to $\text{Set}_i\text{Cell}_0$ if Set_0 moved right. We also calculated distances for Set_0 fishing up, left, and down. These calculations gave us 4 distances for Set_0 , one each for moving right, left, up, and down. The direction with the smallest distance between sets should have the greatest probability, so we divided each of the 4 distances by the smallest distance. Next, we normalized the transformed distances to obtain a probability of Set_0 moving in each direction. We computed these probabilities for each of the sets to determine the direction of fishing. Subsequent set clumps were placed to prevent the overlapping of sets from different clumps ($\text{Seed}+17 \leq x$ or $y \leq \text{Seed}-17$). No constraints were placed upon the overlap of set and turtle clumps, and turtles were distributed as they were in the co-occurrence clumping scenario.

Sets-only clumping scenario In the sets-only clumping scenario, when fishing sets were clumped but sea turtles were uniformly random, the direction of fishing was determined as it was in the independent clumping scenario. When turtles had a uniformly random distribution, they could occur in any cell in the grid of 100×100 cells. To maintain consistency, we placed the same number of turtles across the entire grid in the uniformly random scenarios as we did in the 5 clumps in the clumping scenarios. Distributing 20,250 turtles across the grid with uniform probability resulted in an average of 2.025 turtles/cell and 0.0203 turtles/km². Although turtle densities differed between uniformly random and clumping scenarios, different probabilities of capture were applied in the clumping and uniformly random scenarios to account for higher densities in clumping scenarios. The probabilities of capture are discussed below.

Scenarios with uniformly random sets Fishing sets were uniformly random in 2 spatial scenarios: turtles-only clumping and fully uniform. In these spatial scenarios, set placement and direction of fishing were determined

with a uniform probability distribution. The distributions of sea turtles were constructed as described above.

Simulating bycatch After sea turtles and fishing sets were distributed, we modeled bycatch. To quantify the number of takes, we first tallied the number of turtles that occurred in fished cells. Then, we applied a probability of capture, given co-occurrence of a turtle and set in a cell, to each encountered turtle to determine whether the set caught the turtle (Table 2). The capture probabilities varied across quarter-area strata and were based on observed bycatch rates from the SEFSC for the period from 2005 to 2007 (Walsh and Garrison, 2006; Fairfield-Walsh and Garrison, 2007; Fairfield and Garrison, 2008).

We calculated observed bycatch rates by stratum for leatherback and loggerhead sea turtles for each year from 2005 to 2007 and averaged the stratum bycatch rates across years. We decided to use the rates for leatherback sea turtles in the simulation model because leatherbacks are more of a conservation concern than are loggerheads and the average number of observed sets per year without take was smaller for leatherbacks (100 sets) than for loggerheads (179 sets), and therefore the data for leatherback sea turtles gave us a larger sample size for calculating capture probabilities.

Some of the 40 SEFSC quarter-area strata had no fishing effort, observer coverage, or observed bycatch from 2005 to 2007. The strata without effort or observer coverage were eliminated from the simulation model. Eight strata had either no fishing effort or no observer coverage from 2005 to 2007, and therefore we simulated 32 strata. For observed strata with a bycatch rate of 0 turtles/set, we calculated probabilities of capture from bycatch rates in those strata at different quarters, when possible, or used a median bycatch rate across all quarter-area strata. This algorithm was consistent with the SEFSC's pooling method in which there was pooling across quarters before pooling across fishing areas (Garrison, 2003).

The simulated bycatch probabilities also varied depending on the spatial scenario because of the different turtle densities. The bycatch rates for strata ranged from 0.263 to 0.011 turtles/set. We divided these rates by the average number of sea turtles to be encountered in 5 cells. Then, this probability was applied to each turtles that occurred in fished cells to determine whether it was caught. A set fishing among uniformly random turtles on average encountered 10.125 turtles, and a set fishing among clumped turtles on average encountered 250 turtles.

Observer distribution We attempted to simulate observed fishing sets in a design consistent with the SEFSC's procedure, in terms of both the number of observers and their spatial distribution. The SEFSC's goal for observer coverage has been 8% since 2002 (Beer-kircher et al., 2004). In our model, 8% observer coverage equated to 2 observed sets per computational group of 25 fishing sets.

The SEFSC distributes observers according to a simple random sampling design based on reported effort (Witzell and Cramer, 1995). Vessels are selected for observation in proportion to the amount of fishing reported in a quarter-area stratum in the previous year, and vessels are sampled without replacement within a quarter. Our simulation model was for one year; therefore, we had no effort from a previous year upon which to base the observer distribution. Rather, for each computational group of 25 sets, a single cell from the grid of 100×100 cells was chosen at random to represent an area of high fishing effort. Observers were placed on the 2 sets closest to this cell. Although this method included assumptions differentiating it from the SEFSC's procedure, the most important feature in both practices was the same: observers were distributed independent of bycatch rates.

In summary, the simulation model included the following main assumptions:

- Computational groups of 25 fishing sets distributed across a grid of 100×100 cells where each cell represented 10×10 km
- Clumps of sea turtles and fishing sets represented in grids of 9×9 cells
- Scenarios with clumped turtles in 5 clumps per computational group
- Scenarios with clumped sets in 5 clumps of 5 sets each per computational group
- Overlap of set and turtle clumps
- Methods for placing turtles in the grid
- Methods for placing sets in the grid
- Methods for determining the direction of fishing
- 2 sets selected for observation in each computational group

Properties of estimation methods

We applied 3 estimation methods to the simulated data: 1) the delta-lognormal method, 2) GLM-P, and 3) GLM-NB. Each method was used to estimate bycatch at 2 spatiotemporal scales: 1) for each of the quarter-area strata individually and 2) for all quarter-area strata combined. At the stratum scale, we estimated bycatch for individual quarter-area strata and summed stratum-specific estimates to obtain a total annual bycatch estimate. For the second spatiotemporal scale, we pooled sets across all quarter-area strata and estimated total annual bycatch. Hence, 6 estimates of total annual bycatch (all combinations of the 3 methods and 2 scales) were made for each of the 5 spatial scenarios.

We focused on evaluating the delta-lognormal method because it has long-standing use by the SEFSC. We also chose the GLM-P and GLM-NB because they are simple model-based predictors for count data and, thus, a logical place to begin evaluation of this class of models for estimating bycatch with zero-heavy data. Although the GAM performed well in work reported in McCracken (2004), we did not include it in this analysis because

nonlinear effects of predictor variables have not been extensively studied in the Atlantic and estimation of these effects typically requires data sets larger than the ones available to the SEFSC for annual estimation of bycatch. For example, Kobayashi and Polovina (2005) fit GAMs with 55,785 unobserved sets and 2812 observed sets fished over 5 years in the Pacific fishery. Therefore, in our study, we focused on evaluating the delta-lognormal method and GLMs.

Delta-lognormal estimates are essentially the product of the proportion of fishing sets with bycatch and the average rate of bycatch for those sets (Yeung, 2001). The delta-lognormal method accommodates a predominant group of observations with a value of zero by including a probability of zero catch, and observations with non-zero values are assumed to be lognormally distributed (Pennington, 1983; Ortiz et al., 2000; NMFS, 2001; Fairfield and Garrison, 2008). A lognormal distribution is a continuous probability distribution where the logarithm of the random variable has a normal distribution. Minimum-variance unbiased estimators of means and variances are provided under the delta-lognormal method when data contain many zeros and the non-zero values are lognormally distributed (Pennington, 1983; NMFS, 2001; Garrison, 2003).

The GLM extends the classical linear model by supporting the use of distributions other than the normal distribution. The GLM most commonly applied to count data, the log-linear model, uses a Poisson error distribution (McCullagh and Nelder, 1989). In a Poisson model, counts are assumed to be independent and randomly distributed in space, and the mean and variance of the random variable are assumed to be equal (McCracken, 2004; Sileshi, 2006). However, bycatch data do not always show this relationship. The variance is often larger than the mean—a case known as overdispersion (McCracken, 2004; Potts and Elith, 2006). Patchy distributions, hierarchical data, the observation of a rare event, or lack of independence can lead to the presence of excess zeros, variance heterogeneity, and in turn, overdispersion (McCracken, 2004; Lindsey, 2004; Fahrmeir and Echavarría, 2006). Poisson models are the most commonly used and most straightforward models for count data, but the Poisson distribution accounts for neither zero-inflation nor overdispersion. If overdispersion is not addressed, standard errors can be seriously underestimated and the form of the linear predictor can be misinterpreted (Rideout et al., 2001; Potts and Elith, 2006).

Modeling responses as a negative binomial random variable may be more appropriate if data are overdispersed (Welsh et al., 1996; Thurston et al., 2000; Lindsey, 2004; Venables and Dichmont, 2004). Unlike the Poisson distribution, which has 1 parameter, the negative binomial distribution has 2 parameters: a mean and a dispersion parameter (White and Bennetts, 1996). The dispersion parameter can be understood as a measure of the degree of clumping in a population. The negative binomial distribution with a dispersion parameter that approaches infinity is consistent with

the Poisson distribution where spatial independence is assumed. The spatial independence assumption is relaxed in the negative binomial distribution (White and Bennetts, 1996).

Estimation methods applied

Generalized linear model In the past, the SEFSC and PIFSC have used fishing area, data source (observer or logbook), light stick use, gear depth, month, latitude, sea-surface temperature, day of the year, and number of hooks as explanatory variables in GLMs (Witzell and Cramer, 1995; McCracken, 2004). Our set of potential explanatory variables consisted of all variables recorded both by SEFSC observers and in SEFSC logbooks. It is important that data are recorded in both sources because the model must be fitted with data from observed sets, and data from logbooks must be used to predict bycatch on unobserved sets. The common variables are set number (the sequence of sets within the trip), mainline length, target species, presence of light stick, number of hooks, date, latitude, longitude, sea-surface temperature, and fishing area.

We included mainline length and number of hooks as potential covariates because they are measures of fishing effort and we suspected a positive relationship between amount of effort and number of sea turtles caught. SEFSC data indicate that bycatch rates vary seasonally and spatially. Therefore, we included date as a seasonal covariate and latitude, longitude, and fishing area as spatial covariates. Sea-surface temperature was expected to influence the distribution of sea turtles because they are ectotherms, and research has shown a relationship between temperature gradients and aggregation of sea turtles and swordfish (Bigelow et al., 1999; Polovina et al., 2000; Lewison et al., 2004). Set number, target species, and light stick presence were included in the GLMs as covariates describing fishing methods that may have different levels of interactions with turtles. Gear configuration and fishing method vary depending on the target species and location of fishing (Beerkircher et al., 2004). When targeting swordfish, longlines are set overnight at shallow depths (10–100 m), and a light stick is often attached several meters above the hook on every second or third branchline. In contrast, when tuna are targeted, longlines are set at dawn and hauled in at late afternoon or evening. Further, sea turtles are attracted to light sticks (Wang et al., 2007).

In the simulation model, variable values were selected from real fishing sets observed by the SEFSC from 2005 to 2007 and assigned to simulated sets. When a simulated set had bycatch, we assigned variable values from an SEFSC-observed set with bycatch. Likewise, when a simulated set did not have bycatch, variable values from an SEFSC-observed set without bycatch were assigned.

Variable assignment was also designed to reflect the spatial distribution of simulated fishing sets. In scenarios with uniformly random sets, if the first simulated set in a stratum did not have bycatch, one SEFSC-observed set that did not have bycatch was selected at random

to represent the first simulated set. An analogous procedure was used if the first simulated set in a stratum had bycatch. In scenarios with clumped sets, we assigned the 5 simulated sets in a clump with attributes from SEFSC-observed sets from a single trip. Therefore, SEFSC-observed trips with fewer than 5 sets were eliminated from consideration. The remaining trips were sorted into 6 groups: 0 sets with take, at least 1 set with take and 4 sets without take, at least 2 sets with take and 3 sets without take, at least 3 sets with take and 2 sets without take, at least 4 sets with take and 1 set without take, and at least 5 sets with take. For the first clump in a stratum, an SEFSC-observed trip that had at least as many sets with and without take as the sets simulated in the clump was selected at random. SEFSC-observed sets from that trip were also selected at random to match the simulated number of sets with and without take.

Also, variable values for additional simulated sets in a stratum were selected from SEFSC-observed sets that occurred close in time and space to each other. We calculated the distance from the SEFSC-observed set chosen to represent the first simulated set in a stratum to all other SEFSC-observed sets, indexed by s in the following equation. Because SEFSC-observed sets that were closer to the first-selected set should have a greater probability of selection, the reciprocal of the distance formula was used.

$$Distance_s = \frac{1}{\sqrt{(Date-Date_s)^2 + (Lat-Lat_s)^2 + (Long-Long_s)^2}}.$$

In scenarios with uniformly random sets, the distance values were used to calculate probabilities of selection for each SEFSC-observed set. These probabilities were used to select SEFSC-observed sets to represent the remaining simulated sets within a stratum. The same set could not be selected multiple times within a stratum. The probabilities of selection were recalculated from the first set in subsequent strata.

In scenarios with clumped sets, the average distance value for SEFSC-observed sets within a trip was calculated, and probabilities of selection were calculated for each SEFSC-observed trip. We assigned variable values to the second clump of simulated sets in a computational group by tallying the number of simulated sets with take in the clump and using the calculated probabilities to select an SEFSC-observed trip with the corresponding number of sets with take. Once an SEFSC-observed trip was selected, the required numbers of sets with take and sets without take were selected randomly from the trip. This algorithm continued through all set clumps until the next stratum was reached. The same trip could not be selected multiple times in the same stratum, and the probabilities were recalculated for the next stratum.

To select the best-fitting GLM, we first fit a saturated model and then performed a stepwise procedure based on Akaike information criterion (AIC) values. The re-

sulting model was used to predict the number of sea turtles caught on unobserved sets. The GLMs were fitted in R software, vers. 2.14.1 with the glm2 and glm.nb packages (R Development Core Team, 2011). The glm2 package was used because some models that fail to converge with the glm package may have greater stability with glm2. The glm.nb package is a modification of the glm package with an additional parameter for a negative binomial GLM. We built the simulation model in Microsoft Visual Basic 2010 Express (Microsoft Corp., Redmond, Washington).

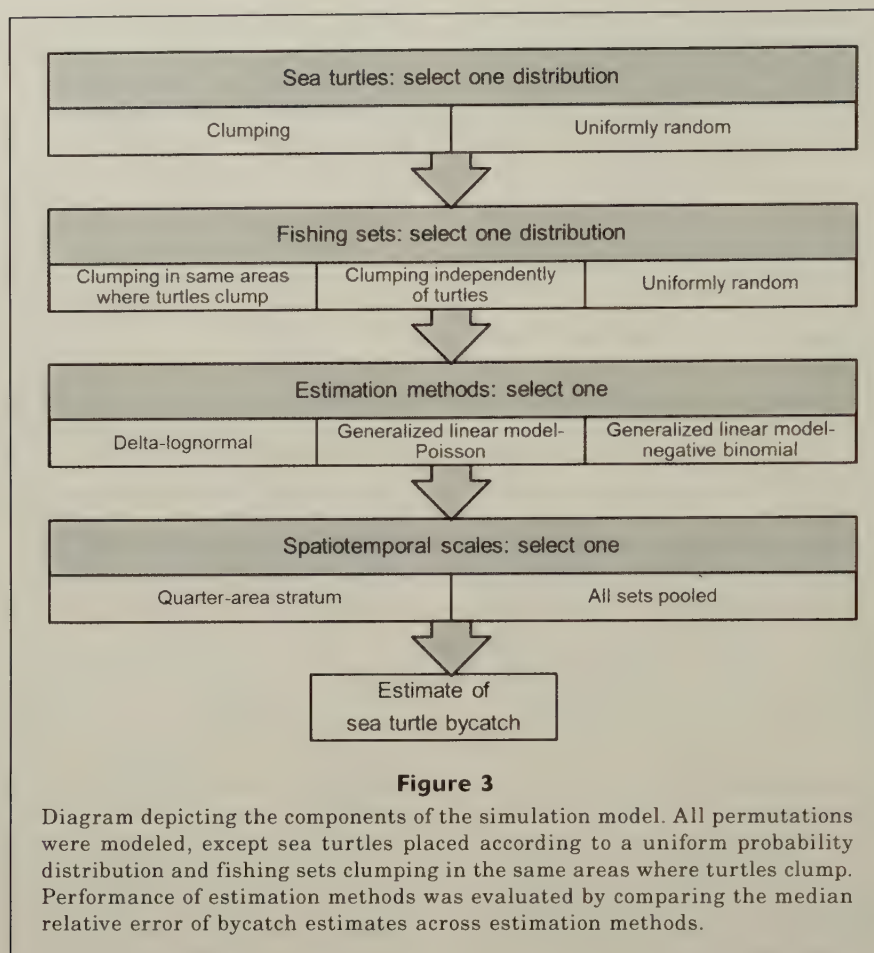
Delta-lognormal method In contrast, the only information besides the observed bycatch required for the delta-lognormal estimation method was the number of hooks per set. The number of hooks per set was simulated according to the procedure we used to assign explanatory variable values to sets for the GLM. To estimate bycatch on unobserved sets, we multiplied the mean observed bycatch rate by the total simulated number of hooks.

Estimating bycatch at 2 spatiotemporal scales The SEFSC estimates bycatch in each quarter-area stratum and sums the estimates across strata to obtain a total annual estimate. In 1999, the SEFSC investigated how pooling data across strata before estimation affects the bycatch estimate (Yeung, 1999). Bycatch point estimates were relatively insensitive to pooling, but estimate precision improved considerably. The only pooling currently done by the SEFSC occurs when a stratum has no observed sets. If a stratum has no observed sets, then the mean bycatch rate of that stratum from previous years is used. Pooling data obscures variation among strata, but it increases the sample size on which bycatch estimates are made. Thus, pooling data addresses the problem of little or no observer coverage and wide confidence intervals. To evaluate the efficacy of a pooling procedure, we pooled simulated data across all quarter-area strata first and then made an estimate of total annual bycatch. We compared this procedure to estimating bycatch in each stratum and summing estimates across strata to obtain a total annual estimate.

Evaluating estimation method performance

An estimation method performs well if point estimates are unbiased and precise. The performance of each estimation method was evaluated under each spatial scenario. Incorporating the different potential sea turtle distributions, set distributions, estimation methods, and spatiotemporal scales of estimation produced 30 potential models: 5 spatial scenarios with 6 estimates each (Fig. 3). Each of the 30 potential models was simulated 1000 times.

We assessed the accuracy of an estimation method by estimating bycatch in 1000 simulations of each spatial scenario, calculating the relative error for each simulation, and identifying the median relative error for the estimation method in that spatial scenario. If an esti-



mation method is unbiased, the median relative error should be zero. The precision of an estimation method can be measured by examining the interquartile range (IQR) of its relative errors. If the IQR of an estimation method is small, then that estimation method is precise.

In addition to evaluating estimation methods using point estimates, we also examined confidence intervals (CIs). After we determined whether the delta-lognormal method or GLMs generated better point estimates, we eliminated the less suitable method from further consideration. Then we analyzed the effects of data pooling on the CIs of the more suitable estimation method. We calculated 95% CIs for each of the 1000 simulations under every spatial scenario and data pooling method. We examined the number of times the simulated total bycatch fell outside the 95% CI for that simulation. We also considered the median CI width for pooling methods under each spatial scenario.

Essentially, we generated 1000 estimates, as the SEF-SC would, for each estimation method in each spatial scenario. We made point estimates with each estimation method for each of the 1000 simulations under every spatial scenario. We calculated 95% CIs for the more suitable estimation method in each of the 1000 simulations under every spatial scenario. Because we knew the total amount of bycatch in each of the 1000 simulations,

we were able to compare the bycatch estimates to the total amount of bycatch simulated and thus evaluate the performance of the estimation method.

Results

Method suitability based on point estimates

We found the delta-lognormal method with stratum-level estimation to be the most accurate of the methods evaluated (Fig. 4). In the co-occurrence clumping scenario ($Turtles_{clump}$, $Sets_{clump-turtles}$) and sets-only clumping scenario ($Turtles_{uniform}$, $Sets_{clump-sets}$), stratum-level estimates were slightly more accurate than pooled estimates, and, in the remaining 3 spatial scenarios, no substantial difference was seen in accuracy between estimates at the stratum-level and estimates from all sets pooled. For each of the 5 spatial scenarios, there was also no substantial difference in the precision between delta-lognormal estimates at the stratum-level and delta-lognormal estimates from all sets pooled.

The GLMs never outperformed the delta-lognormal methods. The GLMs had an accuracy similar to that of the delta-lognormal methods in the fully uniform scenario ($Turtles_{uniform}$, $Sets_{uniform}$), and no substantial

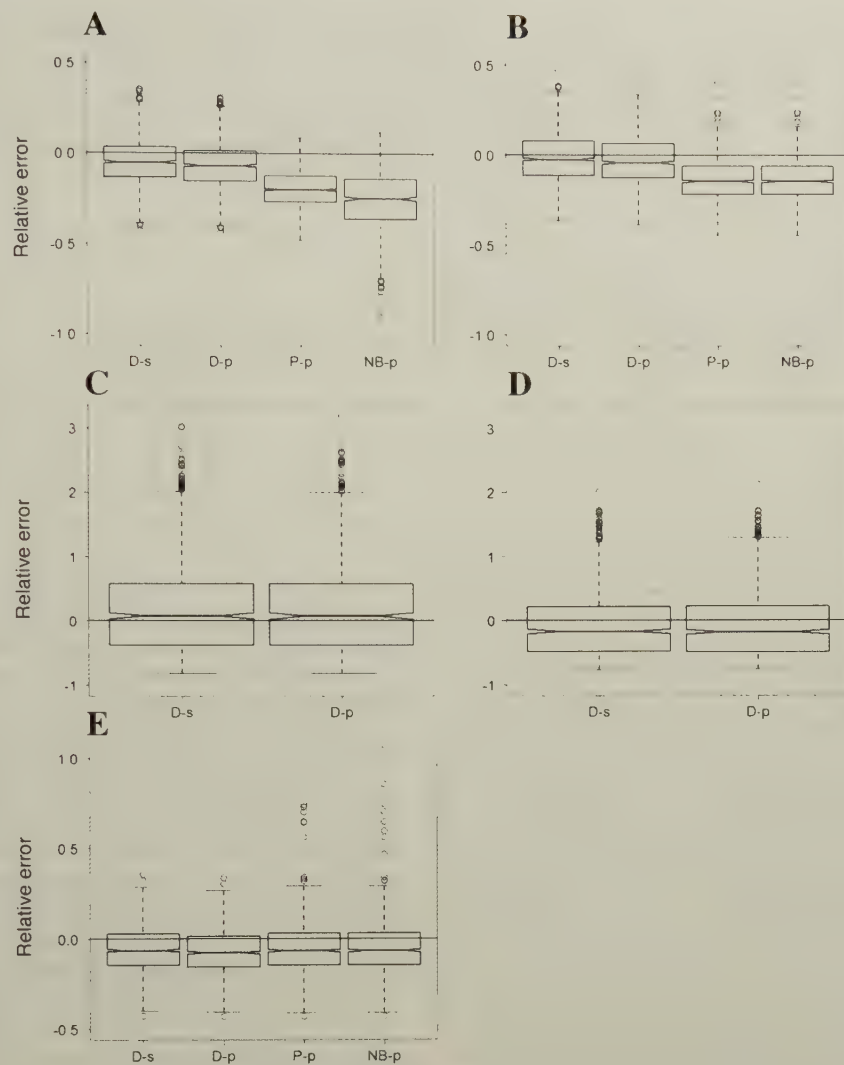


Figure 4

Comparison of bycatch estimates to the total amount of bycatch simulated to evaluate performance of estimation methods. The stratum-level delta-lognormal method (D-s), delta-lognormal method for all sets pooled (D-p), generalized linear model with Poisson error distribution for all sets pooled (P-p), and generalized linear model with negative binomial error distribution for all sets pooled (NB-p) were evaluated. Each of the 5 panels corresponds to one of the spatial scenarios: (A)=co-occurrence clumping ($Turtles_{clump}$, $Sets_{clump-turtles}$), (B)=sets-only clumping ($Turtles_{uniform}$, $Sets_{clump-sets}$), (C)=independent clumping ($Turtles_{clump}$, $Sets_{clump-sets}$), (D)=turtles-only clumping ($Turtles_{clump}$, $Sets_{uniform}$), and (E)=fully uniform ($Turtles_{uniform}$, $Sets_{uniform}$). Each of the plots within a panel corresponds to an estimation method. The scale of the y-axes varies by rows of panels for display purposes. The horizontal line at a relative error of zero marks where the median of an unbiased estimation method should fall. Notches are placed around the medians, and if the notches of 2 plots do not overlap, there is strong evidence that those medians differ. The box of each plot includes the first through third quartile. Whiskers extend to the most extreme data point that is no more than 1.5 times the interquartile range from the box. Small circles represent outliers. For purposes of display, in the panel for the sets-only clumping scenario ($Turtles_{uniform}$, $Sets_{clump-sets}$), one outlier was removed from each of the P-p and NB-p box plots.

Table 3

Median widths of confidence intervals (CIs) from the 5 spatial scenarios and 2 spatiotemporal scales of delta-lognormal estimation in our simulation model of interactions of sea turtles with the U.S. Atlantic pelagic longline fishery. The numbers in parentheses represent the median widths of the CIs as percentages of the bycatch point estimates. The co-occurrence clumping scenario and sets-only clumping scenario were considered the most realistic spatial scenarios.

Spatial scenario	Spatiotemporal scale for estimation	
	Stratum level	All sets pooled
Co-occurrence clumping ($Turtles_{clump}$, $Sets_{clump-turtles}$)	649.8 (84.1%)	402.8 (53.4%)
Independent clumping ($Turtles_{clump}$, $Sets_{clump-sets}$)	100.4 (315.2%)	88.5 (268.7%)
Sets-only clumping ($Turtles_{uniform}$, $Sets_{clump-sets}$)	570.6 (92.5%)	355.7 (59.3%)
Turtles-only clumping ($Turtles_{clump}$, $Sets_{uniform}$)	84.4 (402.6%)	74.0 (322.7%)
Fully uniform distribution ($Turtles_{uniform}$, $Sets_{uniform}$)	523.3 (89.8%)	335.2 (58.0%)

difference was seen between GLM-P and GLM-NB performance (Fig. 4). However, the GLMs produced more outliers than the delta-lognormal methods in the fully uniform scenario ($Turtles_{uniform}$, $Sets_{uniform}$). The GLMs were biased lower than the delta-lognormal methods in the co-occurrence clumping scenario ($Turtles_{clump}$, $Sets_{clump-turtles}$) and sets-only clumping scenario ($Turtles_{uniform}$, $Sets_{clump-sets}$). The GLM-P was less biased and more precise than the GLM-NB in the co-occurrence clumping scenario ($Turtles_{clump}$, $Sets_{clump-turtles}$).

The delta-lognormal method with stratum-level estimation and the delta-lognormal method for all sets pooled performed equally well in the independent clumping scenario ($Turtles_{clump}$, $Sets_{clump-sets}$) and turtles-only clumping scenario ($Turtles_{clump}$, $Sets_{uniform}$). However, in these spatial scenarios, the simulated bycatch rates on sets with observers were much lower than the rates reported to the SEFSC by observers. Although the mean bycatch rate from SEFSC observer data was 0.062 turtles/set (minimum 0.031 turtles/set, maximum 0.081 turtles/set), the mean bycatch rate from simulated observers was 0.006 turtles/set in the independent clumping scenario ($Turtles_{clump}$, $Sets_{clump-sets}$) and 0.004 turtles/set in the turtles-only clumping scenario ($Turtles_{clump}$, $Sets_{uniform}$). By comparison, the mean bycatch rate from simulated observers was 0.122 turtles/set in the co-occurrence clumping scenario ($Turtles_{clump}$, $Sets_{clump-turtles}$), 0.098 turtles/set in the sets-only clumping scenario ($Turtles_{uniform}$, $Sets_{clump-sets}$), and 0.095 turtles/set in the fully uniform scenario ($Turtles_{uniform}$, $Sets_{uniform}$). There were also more outliers in the independent clumping scenario ($Turtles_{clump}$, $Sets_{clump-sets}$) and turtles-only clumping scenario ($Turtles_{clump}$, $Sets_{uniform}$), and the IQRs and whiskers (data within 1.5 times the IQR) were larger for these 2 spatial scenarios than for the other 3 scenarios.

Convergence problems in GLMs

The GLM-P and GLM-NB did not converge for stratum-level estimation in any spatial scenario. For example,

in the co-occurrence clumping scenario ($Turtles_{clump}$, $Sets_{clump-turtles}$), the spatial scenario with the greatest mean observed bycatch rate, the median number of strata with observed take was 19 out of 32. For strata with observed take, the median number of sets with take was 2. The stratum-level GLMs could not converge with such small sample sizes.

Therefore, the GLM-P and GLM-NB methods were considered for estimation only with all sets pooled. Further, for a reason similar to that for the failure of the GLMs at the stratum-level, the GLM-P and GLM-NB methods for estimation with all sets pooled did not converge in the independent clumping scenario ($Turtles_{clump}$, $Sets_{clump-sets}$) or turtles-only clumping scenario ($Turtles_{clump}$, $Sets_{uniform}$). The average number of observed sets with take, out of all observed sets pooled, was 2.64 for the independent clumping scenario ($Turtles_{clump}$, $Sets_{clump-sets}$) and 1.86 for the turtles-only clumping scenario ($Turtles_{clump}$, $Sets_{uniform}$). Therefore, GLM results are not presented for these 2 spatial scenarios.

Confidence intervals

In addition to generating an accurate point estimate, a bycatch estimation method should be able to produce a suitable measure of uncertainty, such as a CI. For every spatial scenario, the median 95% CI calculated from the delta-lognormal method was narrower with estimation from all sets pooled than with estimation from strata (Table 3). In the 2 spatial scenarios thought to be most realistic, the co-occurrence clumping scenario ($Turtles_{clump}$, $Sets_{clump-turtles}$) and sets-only clumping scenario ($Turtles_{uniform}$, $Sets_{clump-sets}$), the median widths of the CIs based on all sets pooled were ~54% and ~59% of the point estimates, respectively. However, the median widths of the CIs from stratum estimates were ~84% and ~93% of the point estimates, respectively (Table 3).

Although the median CIs from all sets pooled were narrower, instances of the total simulated bycatch falling outside the CI occurred more often with all sets pooled than at the stratum level (Table 4). With 95%

Table 4

Number of simulations representing interactions of sea turtles with the U.S. Atlantic pelagic longline fishery in which the simulated amount of bycatch fell outside the 95% confidence interval (CI). We ran 1000 simulations for each of the 5 spatial scenarios and 2 spatiotemporal scales of delta-lognormal estimation. Underestimation occurs when the total simulated amount of bycatch falls above the CI, and overestimation occurs when the total simulated amount of bycatch falls below the CI. The co-occurrence clumping scenario and sets-only clumping scenario were considered the most realistic spatial scenarios.

Spatial scenario	Spatiotemporal scale for estimation			
	Stratum level		All sets pooled	
	Underestimate	Overestimate	Underestimate	Overestimate
Co-occurrence clumping ($Turtles_{clump}, Sets_{clump-turtles}$)	14	0	61	8
Independent clumping ($Turtles_{clump}, Sets_{clump-sets}$)	2	55	2	77
Sets-only clumping ($Turtles_{uniform}, Sets_{clump-sets}$)	1	1	15	10
Turtles-only clumping ($Turtles_{clump}, Sets_{uniform}$)	0	1	0	15
Fully uniform ($Turtles_{uniform}, Sets_{uniform}$)	3	0	60	2

CI's from each of 1000 simulations, it was expected that the total simulated bycatch would fall below the CI in 25 simulations and be above the CI in 25 simulations. The stratum-level CI's for the more realistic spatial scenarios had far fewer than 25 estimates above and 25 estimates below; therefore the stratum-level CI's were too conservative (Table 4). Alternatively, the CI's from all sets pooled performed well in the sets-only clumping scenario ($Turtles_{uniform}, Sets_{clump-sets}$) but, in the co-occurrence clumping scenario ($Turtles_{clump}, Sets_{clump-turtles}$), they contained values that were less than the true amount of bycatch more often than expected (Table 4).

Discussion

Performance of the estimation methods

The delta-lognormal method with stratum-level estimates was the most suitable method in the most realistic spatial scenarios, the co-occurrence clumping scenario ($Turtles_{clump}, Sets_{clump-turtles}$) and sets-only clumping scenario ($Turtles_{uniform}, Sets_{clump-sets}$). This result was seen because observed sets were representative of unobserved sets, sample sizes of observed bycatch were sufficient for estimating bycatch within strata, and model assumptions were not violated.

Observed fishing sets were representative in the co-occurrence clumping scenario ($Turtles_{clump}, Sets_{clump-turtles}$) because all sets fished where sea turtles were present. Likewise, observed sets were representative in the sets-only clumping scenario ($Turtles_{uniform}, Sets_{clump-sets}$) because each set had the same probability of encountering a turtle when turtles had a uniformly random distribution. Further, because these 2 spatial scenarios had enough observed bycatch within strata to make stratum-level estimates, strata did not have to be pooled

to achieve larger sample sizes. Therefore, differences between strata could be captured and potential biases associated with pooling were avoided.

On the other hand, the GLMs could be used only to estimate bycatch for all sets pooled because of convergence problems related to the small amount of observed bycatch in strata. Moreover, the relationship between environmental and fishing conditions and the amount of bycatch was probably not well established in these models because bycatch was rare and observer coverage was low. The use of poorly fitted models could explain why the GLM estimates had lower precision than the delta-lognormal estimates.

The GLMs were as accurate as the delta-lognormal methods in the fully uniform scenario ($Turtles_{uniform}, Sets_{uniform}$) because this spatial scenario was the only one that did not violate the GLM-P assumption that counts are independent and randomly distributed in space (McCracken 2004, Sileshi 2006). Violations of GLM-P assumptions introduced biases in the other spatial scenarios. Additionally, it is likely that the GLM-NB did not perform better than the GLM-P because overdispersion was not a problem (White and Bennetts, 1996; Sileshi, 2006).

In the 2 scenarios where sea turtles were clumped but sets did not mimic their clumping pattern, a low level of bycatch was seen. Under the independent clumping scenario ($Turtles_{clump}, Sets_{clump-sets}$) and turtles-only clumping scenario ($Turtles_{clump}, Sets_{uniform}$), some sets were not expected to encounter any turtles, whereas other sets were expected to encounter many turtles, but the overall frequency of encountering turtles was low. The lowest mean observed bycatch rate occurred in the turtles-only clumping scenario ($Turtles_{clump}, Sets_{uniform}$) with 0.004 turtles/set. This low observed bycatch rate is likely related to the delta-lognormal method having the most bias in this spatial scenario as well. The delta-lognormal method of estimating stratum-level bycatch

had a median relative error of -0.17 in the turtles-only clumping scenario ($Turtles_{clump}$, $Sets_{uniform}$). The median relative error was only -0.05 in the co-occurrence clumping scenario ($Turtles_{clump}$, $Sets_{clump-turtles}$) and -0.02 in the sets-only clumping scenario ($Turtles_{uniform}$, $Sets_{clump-sets}$), the 2 most realistic spatial scenarios.

Confidence intervals

CIs were narrower for estimates from all sets pooled than for stratum-level estimates because the variance in bycatch rates was larger when calculated for strata than when calculated for all sets pooled. Consideration of CI width as a percentage of the bycatch point estimate highlighted how wide and, therefore, uninformative was the standard CI based on strata. Narrowing the CI with calculations from all sets pooled helped address this problem, but the problem of wide CIs was compounded by more underestimation than desired. For protected species conservation, underestimation is more problematic than overestimation. It is important to know a lower bound estimate for protected resource conservation because protected species have an incidental take limit that, if crossed, triggers formal consultation under section 7 of the ESA.

Simulation model assumptions and their implications

Although a simulation model never captures reality perfectly, it is important to consider the effects of model assumptions on results. We attempted to make reasonable assumptions both when incorporating well understood aspects of interactions of sea turtles and the U.S. Atlantic pelagic longline fishery and when modeling unknown features. However, each component of the simulation model could be designed in many ways. We consider the most influential assumptions to be: 1) spatial constraints, 2) the algorithm for selecting explanatory variable values for the GLM, and 3) the simplified effort-based distribution of observers.

First, density of sea turtles, the spatial configuration of sea turtles, the spatial characteristics of fishing, and their interactions had to be defined explicitly in our model. We made assumptions regarding the number of turtles, size of clumps, number of clumps, how turtles or fishing sets should be placed in clumps, how clumps could overlap, and the extent of the study area. Clump placement, the number of turtles per cell, the initial cell of a set, and the direction of fishing had stochastic elements, but model results could be influenced by constraints on the dimensions and spatial distribution of turtles and sets. Further, we acknowledge that our 5 spatial scenarios did not fully replicate reality. We attempted to represent a range of possible distributions, both to model longline interactions with sea turtles and to highlight properties of the estimation methods that could be relevant to other systems. Perhaps, the next step would be to combine multiple spatial scenarios in one model of fishery interactions. In other words, variation in spatial distributions could be more realistically

captured by including more than one spatial scenario in a simulation.

Second, the GLM is based on the premise that environmental or fishing conditions can be used to predict the number of sea turtles caught. Therefore, the manner in which explanatory variable values were assigned to fishing sets in the simulation could have affected GLM performance. We selected variable values from sets observed by the SEFSC from 2005 to 2007 while attempting to account for the spatial distribution and stratum characteristics of the sets. However, variable values could be assigned in many ways, and different procedures could influence how well the GLMs estimated bycatch. Nevertheless, violation of GLM model assumptions could still be a problem even if a more realistic algorithm for selecting explanatory variable values was identified. Since some degree of set and turtle clumping seems to occur in nature and counts are at least dependent within a trip, violations of GLM-P assumptions are likely even with an improved algorithm for selecting explanatory variable values. Perhaps, GLM-NB performance would be improved under a more suitable algorithm for selecting explanatory variable values. However, the GLM-NB is typically used to address overdispersion (Welsh et al., 1996; Thurston et al., 2000; Lindsey, 2004; Venables and Dichmont, 2004), and little overdispersion was detected in our simulation model.

Overall, we do not expect the performance of the GLM-P to change in comparison with the delta-lognormal method. Also, the performance of the stratum-level delta-lognormal method compared with the performance of the delta-lognormal method with all sets pooled is likely robust. However, the GLM-NB could improve its performance relative to the other estimation methods if a clearer functional relationship between the explanatory variables and the level of bycatch was captured.

Third, we modeled a simplified effort-based distribution of observers to simulate observer data for estimating bycatch. If there are different patterns in SEFSC observer data and simulated observer data, the performance of the estimation method in the simulation may not accurately reflect the performance of the estimation method in the actual fishery. The SEFSC currently selects vessels for each calendar quarter and fishing area based on how many sets a vessel fished in that stratum in the previous year (Beerkircher et al., 2004; Fairfield and Garrison, 2008). Vessels that fished more sets in the previous year have a greater chance of being observed by the SEFSC in the current year, and a vessel may be observed up to 4 times a year (Beerkircher et al., 2004). Our simulation model, however, did not cover multiple years, and therefore the quarter-area effort data from the previous year were not available for the distribution of observers. Instead, we selected a cell at random to serve as an area of high effort and placed observers on the 2 sets (of the 25 simulated sets in a computational group) that were closest to that cell. The patterns that we were able to simulate, and that we believe are most relevant, are 8% observer coverage

in each stratum and an observer distribution that is independent of the presence of sea turtles.

Conclusions

Recommendations for management

Bycatch in commercial fisheries is believed to be the main anthropogenic threat to sea turtles, and the pelagic longline fishery is considered one of the 3 fisheries most affecting sea turtles (Witherington et al., 2009). Therefore, improving bycatch estimates is important for sea turtle conservation and effective fishery management. Results from this study indicate that estimating bycatch with the stratum-level delta-lognormal method is appropriate and support the current procedure used by the SEFSC.

General application to zero-heavy data analysis

Not accounting for excess zeros and using models with inappropriate assumptions can result in biased estimates and incorrect conclusions (Martin et al., 2005), as was seen in the performance of the GLMs in our simulation. This study further supports the notion that no one model is clearly most appropriate for analyzing zero-heavy data (Sileshi, 2006). Rather, models must be compared to select a model that is most suitable for the data and the required output (Sileshi, 2006). We cannot recommend one method for addressing all zero-heavy data, but our study shows the importance of recognizing variance across time and space, demonstrates the necessity of representative samples and sample size, and indicates that the delta-lognormal method generates estimates that are less biased and more precise than the GLMs in the case of sea turtle bycatch by the U.S. Atlantic pelagic longline fishery. Many other fields with zero-heavy data also would benefit from an increased understanding of the delta-lognormal method and GLM.

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Abstract—Errors in growth estimates can affect drastically the spawner-per-recruit threshold used to recommend quotas for commercial fish catches. Growth parameters for sablefish (*Anoplopoma fimbria*) in Alaska have not been updated for stock assessment purposes for more than 20 years, although aging of sablefish has continued. In this study, length-stratified data (1981–93 data from the annual longline survey conducted cooperatively by the Fisheries Agency of Japan and the Alaska Fisheries Science Center of the National Marine Fisheries Service) were updated and corrected for discovered sampling bias. In addition, more recent, randomly collected samples (1996–2004 data from the annual longline survey conducted by the Alaska Fisheries Science Center) were analyzed and new length-at-age and weight-at-age parameters were estimated. Results were similar between this analysis with length-at-age data from 1981 to 2004 and analysis with updated longline survey data through 2010; therefore, we used our initial results from analysis done with data through 2004. We found that, because of a stratified sampling scheme, growth estimates of sablefish were overestimated with the older data (1981–93), and growth parameters used in the Alaskan sablefish assessment model were, thus, too large. In addition, a comparison of the bias-corrected 1981–93 data and the 1996–2004 data showed that, in more recent years, sablefish grew larger and growth differed among regions. The updated growth information improves the fit of the data to the sablefish stock assessment model with biologically reasonable results. These findings indicate that when the updated growth data (1996–2004) are used in the existing sablefish assessment model, estimates of fishing mortality increase slightly and estimates of female spawning biomass decrease slightly. This study provides evidence of the importance of periodically revisiting biological parameter estimates, especially as data accumulate, because the addition of more recent data often will be more biologically realistic. In addition, it exemplifies the importance of correcting biases from sampling that may contribute to erroneous parameter estimates.

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Interdecadal change in growth of sablefish (*Anoplopoma fimbria*) in the northeast Pacific Ocean

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Sablefish (*Anoplopoma fimbria*) are a long lived, commercially important finfish abundant along the upper continental slope in the North Pacific, with catches ranging from 10,000 to 35,000 metric tons (t) in Alaskan waters during the last 2 decades (Hanselman et al., 2010). Using data provided by the NOAA National Marine Fisheries Service (NMFS) annual domestic longline survey, we modeled the sablefish population with statistical catch-at-age split by sex (Hanselman et al., 2006). To estimate fish abundance accurately, age-structured models require several biological parameters, such as growth, maturity, natural and fishing mortality, and annual age or length data, as well as annual abundance estimates and catches (Quinn and Deriso, 1999). Errors in growth estimates can drastically affect the spawner-per-recruit threshold used to recommend quotas for commercial fish catches. Overestimation of growth rates may result in overestimation of biomass and, therefore, recommendation of harvest rates that are too high (Quinn and Deriso, 1999). Conversely, underestimation of fish growth can lead to underutilization of a resource and lost economic yield. Growth parameters for Alaskan sablefish have not been updated

for stock assessment purposes since Sasaki's published research (1985). When age-length conversion matrices were first added to the Alaskan sablefish stock assessment in 1995, they were constructed from data (1981–93) that were collected under a length-stratified sampling scheme. These data were randomized according to the method of Kimura and Chikuni (1987), but they were collected in limited areas and over just a few years and were aggregated in a way that put too much weight on large fish (>66 cm FL). For these reasons, we speculated that size estimates used in the assessment of the sablefish population of Alaska have been too large. Meanwhile, many more sablefish have been aged over a larger geographic area. Additionally, since the last update on sablefish growth rates, significant changes in length-at-age have been discovered for other demersal species, such as Pacific halibut (*Hippoglossus stenolepis*) and other flatfish species in the northeast Pacific Ocean. These changes have caused substantial changes in stock assessment results (Walters and Wilderbuer, 2000; Clark and Hare, 2002). Because both sablefish and Pacific halibut have similar fisheries and are such commercially valuable fishes, a change in the assessment

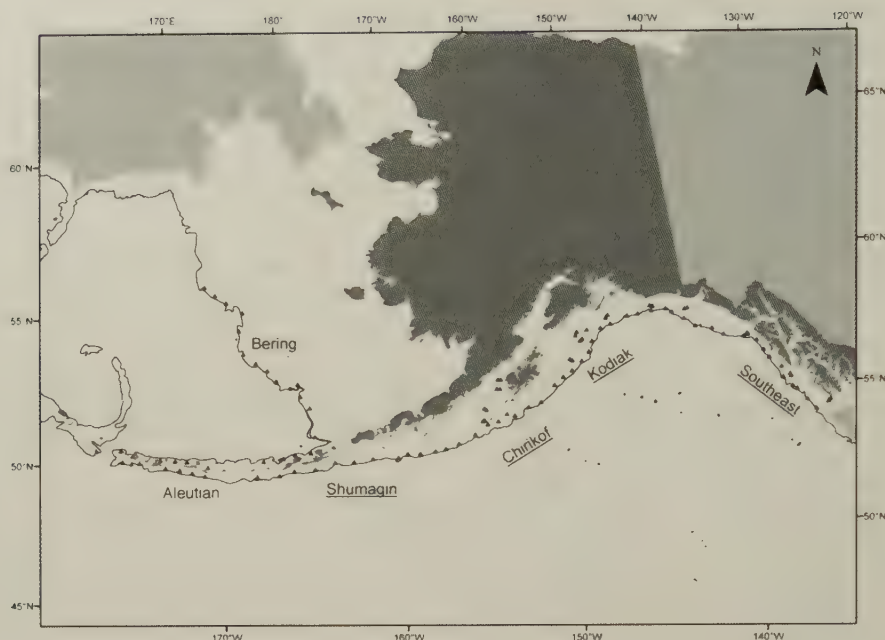


Figure 1

Map showing the 6 management regions covered by the NMFS sablefish (*Anoplopoma fimbria*) longline survey during the period of 1981–2004 and used for growth comparisons in our study: eastern Bering Sea, Aleutian Islands, and the Gulf of Alaska management regions of Shumagin, Chirikof, Kodiak, and Southeast. Underlined regions are those considered to be in the Gulf of Alaska. Triangles represent individual survey stations.

of one of these fishes suggests that an update of the assessment of the other fish is needed.

A growth analysis of sablefish in the Gulf of Alaska (Sigler et al., 1997) revealed values similar to results from the earlier analysis by Sasaki (1985); therefore, the earlier growth estimates for sablefish in Alaska have continued to be used in models for sablefish stock assessment. In the last 20 or more years, however, more sablefish from a wide geographic area have been aged and another evaluation of growth is warranted.

The overall goal for this study was to evaluate whether changes in growth of sablefish in Alaska have occurred since 1985. Specifically, our objectives were 1) to reevaluate estimates of length at age and weight at age, 2) to compare these new estimates among regions and over time for each sex, 3) to evaluate the sensitivity of the current stock-assessment model using this new growth information and evaluate the implications for management of sablefish in Alaska, and 4) to search for biological or environmental reasons for any discovered changes.

Materials and methods

Data collection

We used data available from the annual longline survey conducted cooperatively by the Fisheries Agency of

Japan (1981–94) and the NMFS Alaska Fisheries (AFSC) (1988–present). The Fisheries Agency of Japan conducted the survey solely from 1981 to 1987. Starting in 1988, the NMFS conducted the survey cooperatively with Japan between 1988 and 1994, creating survey overlap between the efforts of the 2 countries. NMFS took over conducting the survey solely in 1995. Samples were collected from June through September, 1981 to 2004, in all 6 management regions defined by the North Pacific Fishery Management Council (NPFMC). Four of these regions are in the Gulf of Alaska (GOA): Southeast, Kodiak, Chirikof, and Shumagin. The other 2 management regions are in the eastern Bering Sea (EBS) and Aleutian Islands (AI) (Hanselman et al., 2010; Fig. 1). Predefined stations have been sampled along the upper continental slope at depths of 200–1000 m in the GOA annually from 1981 to the present and in the BSAI at 2 different schedules annually from 1981 to 1994 and in alternating years from 1996 to the present (EBS in odd years and AI in even years). At each station, 7200 hooks baited with cut squid (*Illex* spp.) and spaced 2 m apart are set (Sigler and Fujioka, 1988).

Length measurements and otoliths of sablefish have been collected since the inception of the Japan-U.S. cooperative longline survey in 1981, and data collection has continued as part of the current NMFS domestic longline survey that started in 1988. However, these data were collected under 2 different sampling designs.

In the first sampling design, fish samples from the Japan-U.S. cooperative survey (1981–93) were stratified by length (5 fish were aged per centimeter length per sex per area). The sex and fork length (FL) of all collected fish were recorded. No assessment of weight was performed.

A change of sampling method took place in 1996 in the NMFS domestic longline survey. A random subsample of fish was collected (if the first hook of a skate contained a sablefish, it was sampled) to acquire age and weight data (Hanselman et al., 2010). A “skate” is a unit of gear that is 100 m long and contains 45 hooks. As before, fork-length measurements and sex of all fish brought aboard were recorded. Age was determined from otoliths stored in 50% ethanol by using the break and burn technique (Beamish and Chilton, 1982; Nielsen and Johnson, 1983).

Length-at-age analysis

Mean length-at-age was calculated from the age-length data in 3 ways by 3 different strata: 1) by sex, region, and survey period, 2) by sex and survey period, and 3) by sex and region. Data were split between the 2 sexes because it was already known that male and female sablefish have different growth rates (Sasaki, 1985) and because the current sablefish assessment model is split by sex. Data were split into 2 periods by using the shift in sampling design: 1981–93 and 1996–2004 (no otoliths were collected in 1994 and 1995). Fish aged 31 years and older were pooled into a 31+ age category (Hanselman et al., 2010). Only the 6 regions sampled consistently across the entire time series (Southeast, Kodiak, Chirikof, Shumagin, EBS, AI; Fig. 1) were used in regional comparisons.

Estimates of mean length-at-age produced by simple averaging with length-stratified data are biased. This bias is caused by aging smaller and larger specimens more often than would be aged under a random sampling design. The mean size-at-age for early age groups is too small, and the mean size-at-age for the oldest age groups is too large (Goodyear, 1995; Sigler et al., 1997; Bettoli and Miranda, 2001). As a result, we determined that size estimates used in the assessment of the sablefish population in Alaska have been too large. To account for stratification, the length-frequency distribution from the survey catch data was used in combination with the length-stratified age samples to create bias-corrected age-length estimates for 1981–93 (Goodyear, 1995; Sigler et al., 1997). The following equation was used (Bettoli and Miranda, 2001):

$$L_a = \frac{\sum_j N_j(n_{a,j}/n_j)l_j}{\sum_j N_j(n_{a,j}/n_j)} \quad (1)$$

Here, L_a = the estimated mean length at age a ;

l_j = the median of the length group j ;

N_j = the number of fish in the j th length group;

n_j = the number of fish subsampled for age determination in the j th length group; and

$n_{a,j}$ = the number of fish in age group a in the subsample from the j th length group.

Sablefish growth was modeled with the von Bertalanffy (VB) age-length model, which was fitted by non-linear least squares weighted by sample size,

$$\bar{L}_a = L_\infty(1 - e^{-\kappa(a - t_0)}) + \varepsilon_a \quad (2)$$

Here, L_∞ = the average maximum length;

κ = the mean growth coefficient;

t_0 = the mean theoretical age a fish would have been at zero length; and

ε_a = an additive normally distributed error term.

Standard errors, correlation estimates, and 95% confidence intervals for growth curve parameters were estimated by the Hessian method of second partial derivatives (Quinn and Deriso, 1999).

Individual parameters of growth models were compared using the univariate Fisher-Behrens test. Likelihood ratio tests (LRTs) were carried out to determine whether growth curves differed between the 2 survey periods, among regions, or both survey period and region (Kimura, 1980; McDevitt, 1990; Sigler et al., 1997). The LRT for comparing nested models was log-transformed and calculated as follows:

$$-2 \ln(RSS_F / RSS_R) \sim \chi^2 \quad (3)$$

Here, N = the total number of observations (of length-at-age); and

RSS_F and RSS_R = the estimated residual sum of squares (RSS) of the full (F) and reduced (R) models, respectively (Kimura, 1980; Quinn and Deriso, 1999).

The degrees of freedom for the test are the difference in the number of parameters between the full and reduced models. The increase in the RSS between each of the reduced models and the full model was used to test for temporal and spatial effects. This increase also was used to further test for differences among pairs of regions and between survey periods within each region if a regional or temporal effect was discovered.

Weight-at-age analysis

Weight-at-age curves were fitted to data by sex and region strata. Sasaki (1985) reported sablefish weight estimates; however, no weight data were collected before 1996 in the domestic longline survey; therefore, no temporal changes were investigated. Because weight data were collected only from random samples, no correction for stratification was needed. Fish of ages >31 were pooled into a 31+ age category (Hanselman et al., 2010). To determine weight-at-age for the stock assess-

ment model, first the length-weight relationship was determined by using the typical nonlinear allometric relationship:

$$\hat{W}_a = \alpha \cdot l + \varepsilon. \quad (4)$$

Here, length l , α , and β are parameters estimated by procedures for nonlinear least squares. This equation was combined with the length-at-age model to construct the weight-at-age model. The weight-at-age model was log-transformed to the following equation because the data had a multiplicative error structure:

$$\ln \hat{W}_a = \ln W_\infty + \beta \cdot \ln(1 - e^{(-k(a-t_0))}) + \varepsilon, \quad (5)$$

where ε = a normally distributed error term.

Because of high parameter correlation with only one dependent variable, the allometric parameter β was fixed, determined from the length-weight relationship. The 3 remaining parameters, W_∞ , k , and t_0 were estimated by a nonlinear procedure (Quinn and Deriso, 1999).

Two age-weight models were fitted to each sex to test whether sablefish weight-at-age differed by region. The full model used separate growth curves fitted to each of the 6 regions, and the reduced model relied on one growth curve fitted to pooled data. Equation 3 was used to compare the full model against the reduced model at a significance level of $\alpha=0.05$ (Sigler et al., 1997; Quinn and Deriso, 1999).

Biological and oceanographic explanations for observed changes

Several hypotheses have been formulated to explain the possible change in growth of sablefish in Alaska: interspecific competition with healthy arrowtooth flounder (*Atheresthes stomias*) populations, intraspecific density-dependent processes, and changing environmental conditions (Hanselman et al., 2006; Maloney and Sigler, 2008). We explored the possibility that temporal growth changes can be attributed to density-dependent effects or to environmental factors, including winter sea-surface temperature (SST), summer SST, and the Pacific Decadal Oscillation (PDO) index.

To test for intra- and interspecific density-dependence, linear regressions were performed between each of the response variables (the growth parameter k , mean length at age 4, and mean length at age 6), and each of the explanatory variables (biomass values for age-2 sablefish, age-4+ sablefish, and age-4+ arrowtooth flounder). Biomass estimates were obtained from the 2008 Alaskan sablefish stock assessment (Hanselman et al., 2007) and 2008 Alaska arrowtooth flounder stock assessment (Turnock and Wilderbuer, 2007). Growth estimates were taken from data pooled across the entire series, 1981–2004, for all regions, fitted to the von Bertalanffy growth curve. Significance was determined using a level of $\alpha=0.05$, and then the coefficients of

determination (r^2) were used to assess the explanatory power of the model.

To discern the effect of density-dependence while sablefish were in the juvenile stage, abundance estimates for sablefish and arrowtooth flounder were lagged by 2, 3, and 4 years. This calculation was made to compare the growth rate and size of sablefish at age 4 and age 6 with the abundance of sablefish and arrowtooth flounder exposed to while young of the year (YOY), and at age 1, age 2, and age 3.

To examine the influence of environment on growth, linear regressions were performed between each of the response variables (mean length at age 4 and mean length at age 6), and each of the explanatory variables (winter SST, summer SST, and an index used to quantify the PDO). Because YOY and juvenile sablefish are more susceptible to surface temperatures and are considered to be more susceptible to oceanographic variability than are adults, we lagged the SST by 2, 3, and 4 years to compare the size of an age 4 sablefish with the SST exposed to while as a YOY, and at age 1, and age 2, and we lagged the SST by 4, 5, and 6 years to compare the size of an age 6 sablefish with the SST exposed to as a YOY, and at age 1 and age 2.

Monthly values of the PDO index were obtained from the Joint Institute for the Study of Atmosphere and Oceans (Mantua et al., 1997; <http://jisao.washington.edu/pdo/PDO.latest>, accessed January 2008; <http://www.beringclimate.noaa.gov/data/index.php>, accessed January 2008), which incorporated data from the United Kingdom's Meteorological Office's (UKMO) Historical SST Dataset and Reynolds' Optimally Interpolated SST. SST values for the Bering Sea (<http://www.beringclimate.noaa.gov/data/BCresult.php>, accessed January 2008) and the GOA (Kaplan et al., 1998; <http://www.esrl.noaa.gov/psd/data/timeseries/>, accessed January 2008) were taken from a data set of SST anomalies, Kaplan Extended SST V2, provided by the Physical Sciences Division of NOAA's Earth System Research Laboratory, Boulder, Colorado.

Management implications

We examined the sensitivity of the current stock assessment model to the use of the new growth information from our study. The AFSC models the Alaskan sablefish population with statistical catch-at-age methods. It uses a penalized maximum likelihood function to estimate parameters simultaneously to obtain the best fit between predicted and observed data. Data in the sablefish stock assessment model include catch, several abundance indices, and age and length data from the longline survey and from the fishery. For details of the assessment model, see Hanselman et al. (2010).

This assessment model in its current form uses age-length conversion matrices, not empirical age-length keys, to describe the probability that a fish of a given age is of a certain length. This model uses these age-length conversion matrices to predict lengths. The weight-at-age is input as a fixed vector for the whole

Table 1

Growth parameters (L_{∞} =average maximum length, κ =mean growth coefficient, t_0 =mean theoretical age a fish would have been at zero length) for male sablefish (*Anoplopoma fimbria*) in Alaska estimated with data from the annual longline survey conducted cooperatively by the Fisheries Agency of Japan and the Alaska Fisheries Science Center of the National Marine Fisheries Service in 1981–93 and by the Alaska Fisheries Science Center during 1996–2004. Estimates were made for 6 management regions with the von Bertalanffy model fitted to age-length data stratified by region and survey period, where n is the number of age-length observations and an asterisk (*) indicates a significant difference between the 2 periods, 1981–93 and 1996–2004, in that particular region. The 6 regions are the Chirikof, Kodiak, Shumagin, and Southeast, all in the Gulf of Alaska, and the eastern Bering Sea and Aleutian Islands. Standard errors of the mean (SE) are provided in parentheses. RSS=residual sum of squares.

Region	Survey period	L_{∞}	κ	t_0	RSS	n
All regions combined	1981–93	64.6 (0.38)	0.287 (0.03)	–2.07 (0.60)	3644	3429
	1996–2004*	67.7 (0.16)	0.292 (0.01)	–2.25 (0.21)	904	2614
	1981–2004	66.2 (0.28)	0.30 (0.03)	–2.19 (0.51)	18,954	6043
Chirikof	1981–93	70.2 (1.02)	0.239 (0.03)	–2.288 (0.70)	448	128
	1996–2004*	67.3* (0.48)	0.335 (0.06)	–1.617 (1.02)	487	294
	1981–2004	67.8 (0.45)	0.327 (0.03)	–1.287 (0.48)	1230	422
Aleutian	1981–93	67.0 (0.55)	0.195 (0.03)	1329	726	
	1996–2004	68.1 (0.48)	0.243 (0.02)	–2.898 (0.59)	478	543
	1981–2004	67.0 (0.55)	0.195 (0.03)	–5.101 (1.23)	2235	1269
Kodiak	1981–93	65.1 (0.66)	0.352 (0.06)	–1.685 (0.79)	1737	598
	1996–2004*	66.6 (0.34)	0.357 (0.07)	–2.052 (1.21)	606	542
	1981–2004	66.0 (0.39)	0.365 (0.04)	–1.423 (0.55)	3239	1140
Shumagin	1981–1993	64.3 (0.50)	0.440 (0.07)	–0.793 (0.60)	1625	684
	1996–2004*	70.1 (0.98)	0.193 (0.03)	–4.501 (1.08)	438	267
	1981–004	65.3 (0.49)	0.352 (0.05)	–1.669 (0.63)	2914	951
Bering	1981–1993	64.9 (0.64)	0.197 (0.04)	–6.264 (1.67)	1154	757
	1996–2004*	69.3* (0.50)	0.237 (0.03)	–3.48 (0.86)	600	363
	1981–2004	66.7 (0.71)	0.186 (0.03)	–6.250 (1.69)	4695	1120
Southeast	1981–1993	67.0 (0.79)	0.219 (0.04)	–3.827 (1.19)	1998	536
	1996–2004*	68.3 (0.37)	0.307 (0.04)	–1.714 (0.73)	829	605
	1981–2004	67.7 (0.45)	0.271 (0.03)	–2.384 (0.65)	4136	1141

time series. If the conversion matrices and the weight-at-age vector are developed with growth data that do not correspond with the true underlying growth, they can bias the stock assessment (Hanselman et al., 2007). Using the updated growth curves from the 2 survey periods reported in this study, we created new length-age conversion matrices and a new weight-at-age vector and applied them to the current stock assessment model (Hanselman et al., 2007).

Results

Length-at-age analysis

Our results indicate that previously used growth estimates in the stock assessments of sablefish in Alaska (assessments before 2007) obtained from length-stratified sampling were erroneously too large. A comparison of growth estimates from 1981–93 data updated to correct for this bias with estimates from more recent data (1996–2004) indicates that sablefish are growing to a larger maximum size in more recent years. The

estimates of average maximum length (L_{∞}) used in the stock assessment of Alaskan sablefish in 2007 (Sasaki's [1985] estimate from length-stratified data) were 69 cm FL for males and 83 cm FL for females (Hanselman et al., 2006). Maximum lengths were smaller in our bias-corrected estimates for the same time period (1981–93; Tables 1, 2) than in the 2007 stock assessment model: males=64.6 cm FL, females=75 cm FL. Our estimates for the more recent period (1996–2004; Tables 1, 2) are significantly larger (males=67.7 cm FL, females=80.1 cm FL) than the bias-corrected estimates from the earlier period, but these estimates for the recent period are still smaller than the estimated lengths incorrectly used in earlier stock assessments.

The growth rates of male and female sablefish in Alaska differed significantly across areas and survey periods ($P<0.05$; Tables 3, 4). In the data from the earlier period, both male and female sablefish display smaller asymptotic lengths (L_{∞}) and younger ages t_0 than do sablefish in data from the more recent time period (Fig. 2). Significant differences were detected between the 2 male growth curves ($P<0.001$; Table 1). Test results on the female data showed that the L_{∞}

Table 2

Growth parameters (L_{∞} =average maximum length, κ =mean growth coefficient, t_0 =mean theoretical age a fish would have been at zero length) for female sablefish (*Anoplopoma fimbria*) in Alaska estimated with the von Bertalanffy model fitted to age-length data stratified by region and time period, where n is the number of age-length observations and an asterisk (*) indicates a significant difference between the 2 survey periods, 1981–93 and 1996–2004, for that particular region. Standard errors of the mean (SE) are presented in parentheses. RSS=residual sum of squares.

Region	Survey period	L_{∞}	κ	t_0	RSS	n
All regions combined	1981–93	75.0 (0.35)	0.263 (0.01)	–2.00 (0.29)	3945	4788
	1996–2004*	80.1* (0.26)	0.223 (0.01)	–1.92 (0.14)	1191	3493
	1981–2004	77.1 (0.77)	0.25 (0.02)	–1.91 (0.32)	23,963	8281
Chirikof	1981–93	75.3 (1.29)	0.298 (0.04)	–0.798 (0.58)	1275	165
	1996–2004	77.5 (0.51)	0.294 (0.02)	–0.802 (0.40)	609	485
	1981–2004	76.8 (0.56)	0.302 (0.02)	–0.697 (0.33)	2201	650
Aleutian	1981–93	73.8 (0.84)	0.197 (0.04)	–3.888 (1.30)	4839	1037
	1996–2004*	77.9* (1.31)	0.218 (0.03)	–2.246 (0.69)	2191	795
	1981–2004	73.8 (0.69)	0.248 (0.03)	–2.210 (0.70)	9466	1832
Kodiak	1981–93	74.5 (0.84)	0.305 (0.04)	–1.288 (0.51)	3334	831
	1996–2004*	78.6* (0.50)	0.311 (0.03)	–0.49 (0.42)	1081	602
	1981–2004	76.7 (0.63)	0.292 (0.03)	–1.220 (0.40)	7231	1433
Shumagin	1981–93	73.2 (0.69)	0.295 (0.03)	–1.724 (0.58)	1993	975
	1996–2004*	81.6* (1.30)	0.177* (0.02)	–3.046 (0.49)	877	563
	1981–2004	74.7 (0.72)	0.256 (0.02)	–2.028 (0.48)	4830	1538
Bering	1981–93	68.3 (0.59)	0.351 (0.05)	–1.79 (0.76)	1925	993
	1996–2004*	76.4* (0.87)	0.223* (0.02)	–2.746 (0.62)	695	533
	1981–2004	70.2 (0.74)	0.306 (0.05)	–2.163 (0.82)	6979	1526
Southeast	1981–93	78.3 (1.02)	0.189 (0.03)	–3.579 (0.95)	4488	787
	1996–2004*	80.8* (0.50)	0.273* (0.02)	–0.816* (0.42)	964	515
	1981–2004	79.3 (0.72)	0.217 (0.02)	–2.489 (0.61)	8949	1302

Table 3

Comparison of 4 age-length models used for analyses of regional and temporal effects on growth of male sablefish (*Anoplopoma fimbria*) in Alaska. The most reasonable model, indicated with an asterisk (*), is the reduced model with a residual sum of squares (RSS) not significantly greater than the RSS for the full model. n =the number of observations, and χ^2 =the chi-squared value.

Model	RSS	χ^2	P	No. of parameters	n
Data split by each combination of region and survey period *	11,729			36	
Data split into 2 survey periods	16,354	114.7	<0.001	6	
Data split into 6 regions	18,449	156.3	<0.001	18	
All data pooled	21,642	211.3	<0.001	3	6043

estimates ($P<0.001$) and the growth curves were significantly different ($P<0.001$; Table 2) between the 2 periods.

A comparison of male growth curves between the 2 survey periods, stratified by region, showed a consistent pattern of slower growth and smaller asymptotic lengths during the earlier survey period. There were significant differences between growth curves fitted to the 2 periods in 5 of the 6 management regions (Fig. 3, Table 1). Fish from most regions had a smaller asymp-

totic length and slower growth during the earlier survey period than during the more recent period. In contrast, during the earlier survey period versus the more recent one, males in the Shumagin region reached a smaller maximum length but grew faster and males in the Chirikof region displayed a larger asymptotic length and grew more slowly.

Age-length relationships for females between the 2 survey periods differed significantly in 5 of the 6 management regions (Fig. 4, Table 2). Female asymptotic

Table 4

Comparison of 4 age-length models used for analyses of regional and temporal effects on growth of female sablefish (*Anoplopoma fimbria*) in Alaska. The most reasonable model, indicated with an asterisk (*), is the reduced model with a residual sum of squares (RSS) not significantly greater than the RSS for the full model. n =the number of observations, and χ^2 = the chi-squared value.

Model	RSS	χ^2	P	No. of parameters	n
Data split by each combination of region and survey period *	24,271			36	
Data split into 2 survey periods	48,717	238.9	<0.001	6	
Data split into 6 regions	39,656	168.4	<0.001	18	
All data pooled	68,900	357.9	<0.001	3	8281

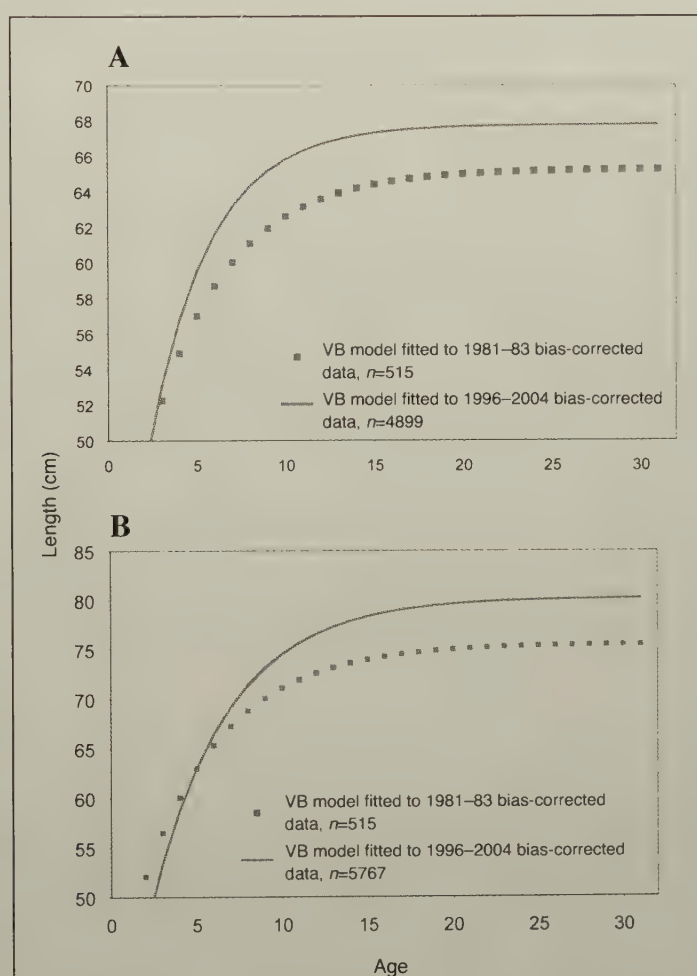
lengths ranged from 68.3 to 78.3 cm FL during the earlier survey period, with the lowest maximum lengths occurring in the EBS region and highest lengths in the Southeast. In the more recent period, asymptotic lengths were much larger, ranging from 76.4 cm (EBS) to 81.6 cm FL (Shumagin region). During the earlier time period, compared to the more recent one, AI, Kodiak, and Southeast females grew slower and Shumagin, Chirikof, and EBS females displayed the opposite pattern.

Several tests for differences in growth between pairs of regions were significant ($P<0.05$) for both sexes. Male sablefish showed fewer differences in growth between regions, with Chirikof males differing significantly from Shumagin ($P=0.02$), AI ($P=0.01$) and EBS ($P=0.01$) males, and EBS males differing significantly from males in the Southeast ($P=0.04$). For female sablefish, most regional comparisons were highly significant, with the exception of the difference between AI and Shumagin ($P=0.55$), Chirikof and Kodiak ($P=0.12$) and Southeast ($P=0.12$), and Kodiak and Southeast ($P=0.37$).

A consistent pattern of smaller estimates of t_0 , the theoretical age at zero length, was seen for both male and female sablefish in the earlier survey period, than estimates for the more recent survey period. These smaller values could be a result of small sample sizes of fish <4 years old in the older data sets (Sigler et al., 1997).

Weight-at-age

The age-weight relationship differed significantly among regions in both males ($P<0.001$, Table 5) and females ($P<0.001$, Table 6). Maximum weights for male (Table 7) and female (Table 8) sablefish in all regions combined were smaller than the values used in the current stock assessment model, likely because of differences in age at length. Female sablefish in pooled regions reached a higher average maximum weight-at-age than did male sablefish, 5.5 kg versus 3.2 kg, respectively.

**Figure 2**

Comparison of (A) male and (B) female sablefish (*Anoplopoma fimbria*) von Bertalanffy (VB) growth curves (for all of the 6 management regions in Alaska combined) fitted to bias-corrected age-length data from 1981 through 1993 (shown as a dashed line) and fitted to age-length data from 1996 through 2004 (shown as a solid line). Note the different scales on the y-axis of each figure. Sample sizes (n) contributing to the VB analysis are listed in the legends.

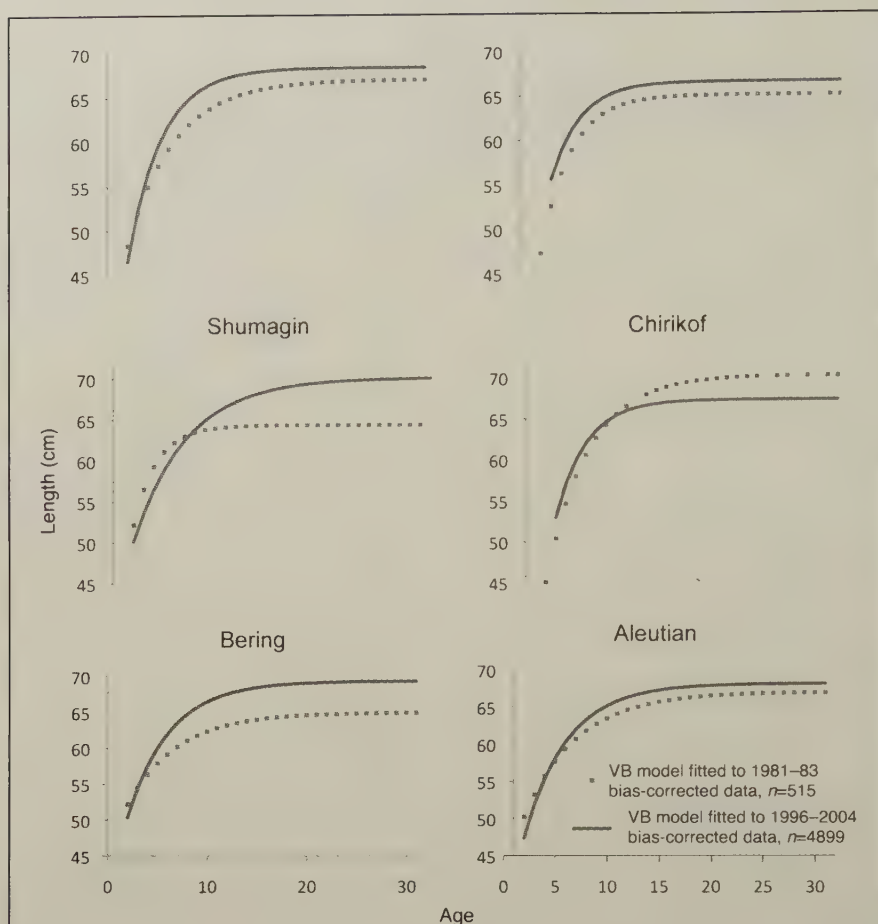


Figure 3

Comparison by management region of male sablefish (*Anoplopoma fimbria*) von Bertalanffy (VB) growth curves fitted to bias-corrected age-length data from the period 1981–93 (shown as a dotted line) and fitted to age-length data from the period 1996–2004 (shown as a solid line). Sample sizes (n) contributing to the VB analysis are listed in Table 1.

Table 5

Comparison of 2 age-weight models used for analyses of regional effects on growth of male sablefish (*Anoplopoma fimbria*) in Alaska. The most reasonable model, indicated with an asterisk (*), is the reduced model with a residual sum of squares (RSS) not significantly greater than the RSS for the full model. n =the number of observations, and χ^2 =the chi-squared value.

Model	RSS	χ^2	P	No. of parameters	n
Data split into 6 regions*	144.4			24	
All data pooled	151.8	174.6	<0.001	4	2614

Maximum average weights among male sablefish varied slightly, but still significantly ($P<0.05$), by region (Table 7). The lightest males, with maximum weight of 3.0 kg, were found in the Kodiak region, and the heaviest males, with maximum weight of 3.4 kg, were observed in the EBS region. Females showed a larger

range of average maximum weights, from 4.7 kg in the EBS region to 5.8 kg in the Shumagin region (Table 8). Several maximum weights differed significantly between regions for both sexes; similar age-weight relationships were seen only for females in the AI and Shumagin regions and the Kodiak and Southeast regions.

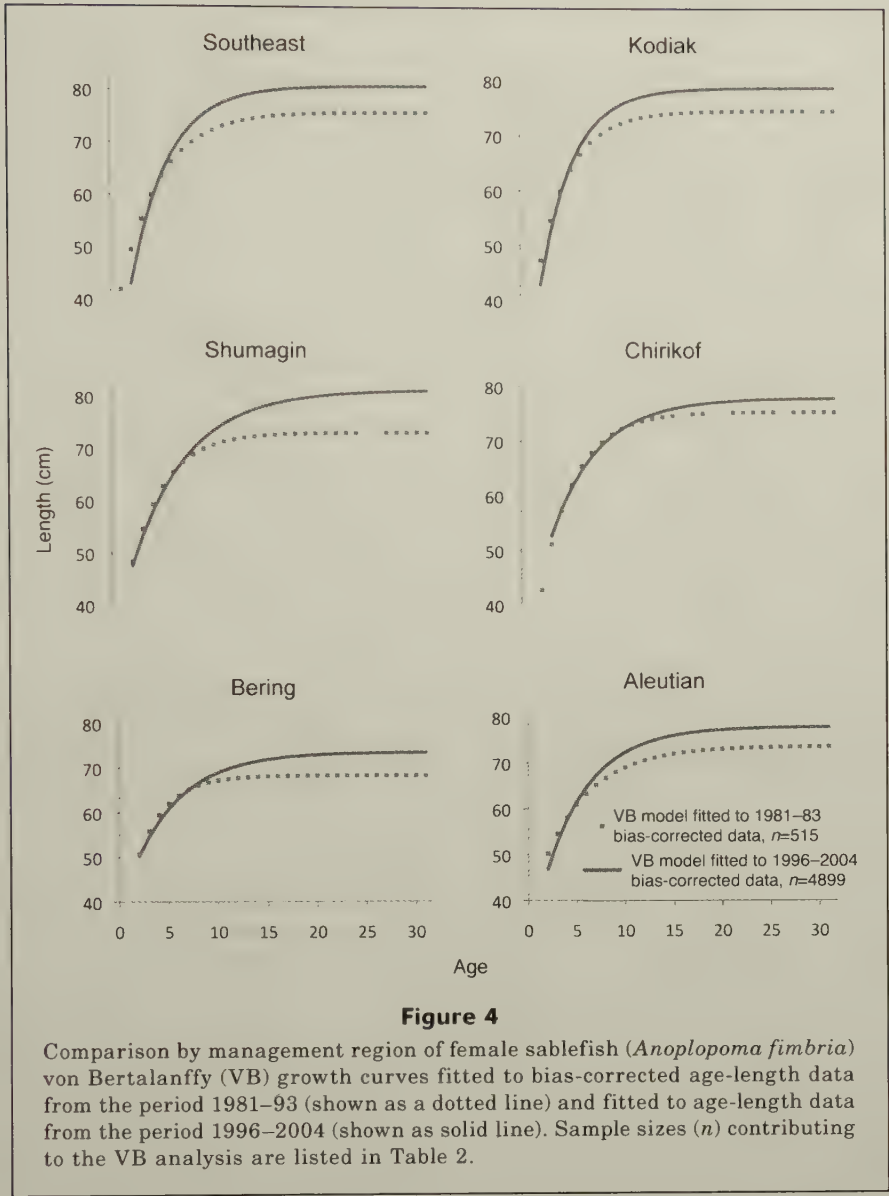


Figure 4
Comparison by management region of female sablefish (*Anoplopoma fimbria*) von Bertalanffy (VB) growth curves fitted to bias-corrected age-length data from the period 1981–93 (shown as a dotted line) and fitted to age-length data from the period 1996–2004 (shown as solid line). Sample sizes (*n*) contributing to the VB analysis are listed in Table 2.

Table 6

Comparison of 2 age-weight models used for analyses of regional effects on growth of female sablefish (*Anoplopoma fimbria*) in Alaska. The most reasonable model, indicated with an asterisk (*) is the reduced model with a residual sum of squares (RSS) not significantly greater than the RSS for the full model. *n*=the number of observations, and χ^2 =the chi-squared value.

Model	RSS	χ^2	<i>P</i>	No. of parameters	<i>n</i>
Data split into 6 regions *	262	145.5	<0.001	24	3493
All data pooled	277			4	

Male sablefish in all of the 6 regions displayed highly significant differences in weight-at-age, although their growth curves appeared similar. These minor growth differences may not be of biological importance and may not need to be considered for assessment purposes.

Biological and oceanographic explanations for observed changes

There was no evidence of a common climatic forcing factor among the management regions in relation to

Table 7

Estimates of weight-at-age parameters (W_{∞} =average maximum weight, κ =mean growth coefficient, t_0 =mean theoretical age a fish would have been at zero weight) for male sablefish (*Anoplopoma fimbria*) in Alaska determined with the von Bertalanffy model fitted to age-weight data for the pooled survey periods of 1996–2004 stratified by region and combined for Alaskan waters (W_{∞} = average maximum weight, κ =mean growth coefficient, t_0 =mean theoretical age a fish would have been at zero weight). Standard errors of the mean (SE) are presented in parentheses. β was fixed at 3 and is, therefore, not included in this table. n =the number of age-weight observations.

	W_{∞}	κ	t_0	RSS	n
All regions pooled	3.2 (0.03)	0.355 (0.01)	-1.113 (0.18)	152	4889
Aleutian	3.3 (0.09)	0.285 (0.03)	-1.949 (0.50)	38.1	543
Bering	3.4 (0.07)	0.313 (0.03)	-1.630 (0.47)	17.4	363
Chirikof	3.1 (0.06)	0.460 (0.07)	0.019 (0.59)	13.9	294
Kodiak	3.0 (0.03)	0.762 (0.10)	1.106 (0.35)	23.2	542
Shumagin	3.3 (0.15)	0.272 (0.04)	-2.252 (0.73)	18.3	267
Southeast	3.2 (0.04)	0.421 (0.03)	0.019 (0.30)	33.5	605

Table 8

Estimates of weight-at-age parameters for female sablefish (*Anoplopoma fimbria*) determined with the von Bertalanffy model fitted to age-weight data for the pooled survey period of 1996–2004 stratified by region and combined for all Alaskan waters (W_{∞} = average maximum weight, κ =mean growth coefficient, t_0 =mean theoretical age a fish would have been at zero weight). Standard errors (SE) are presented in parentheses. β was fixed at 3 and is, therefore, not included in this table. n =the number of age-length observations.

	W_{∞}	κ	t_0	RSS	n
All regions pooled	5.5 (0.06)	0.238 (0.01)	-1.387 (0.13)	277	5767
Aleutian	5.5 (0.22)	0.209 (0.02)	-2.092 (0.37)	71.5	795
Bering	4.7 (0.16)	0.267 (0.02)	-1.598 (0.42)	34.2	533
Chirikof	5.0 (0.12)	0.326 (0.03)	-0.206 (0.33)	29.5	485
Kodiak	5.2 (0.10)	0.336 (0.02)	-0.064 (0.27)	42	602
Shumagin	5.8 (0.33)	0.197 (0.02)	-2.349 (0.37)	47.9	563
Southeast	5.5 (0.11)	0.300 (0.02)	-0.114 (0.27)	38.2	515

changes in sablefish growth. Arrowtooth flounder biomass likewise was unrelated to sablefish growth. Intra-specific, density-dependent effects appeared to be a more plausible explanation for changes in growth of Alaskan sablefish because measures of age-4+ biomass at some lags (in years) were correlated with reduced growth. Significant relationships included mean length at age 6 regressed on the total age-4+ biomass (coefficient of determination [r^2]=0.28, P =0.02), and mean length at age 4 regressed on the age-4+ biomass from 3 years prior, when the sablefish were age 1 (r^2 =0.5, P =0.04). Both of these analyses revealed a decrease in average length with an increase in biomass. Although not significant, a negative correlation between the growth coefficient κ and both age-2 and age-4+ biomass also was evident.

Management implications

The use of updated growth data (length-at-age fitted to 2 survey periods and weight-at-age from the more recent

survey period) improved the fit of the current AFSC assessment model of sablefish to the data and slightly increased the recommended fishing-induced mortality. The updated growth also had an effect on the estimated time series of female spawning biomass (Fig. 5). Three prominent changes in the estimates of female spawning biomass were observed when the assessment model that used the estimates of Sasaki (1985) was run with our updated growth estimates: 1) the initial estimated spawning biomass in 1960 was substantially higher, 2) the minima in female spawning biomass are lower, and 3) the estimated spawning biomass was slightly lower for recent years (2000 to present). The increase, between the use of Sasaki estimates and our bias-corrected data, in estimated spawning biomass in 1960 is biologically reasonable because fishing mortality before 1960 was low (Hanselman et al., 2007). The lower spawning biomass minima in the updated series imply that the resource was not managed as conservatively as expected during the periods of lowest biomass. Results from our study

show that recent estimates of female spawning biomass from our updated growth data are slightly lower but appear to be rising at a steeper trajectory than recent estimates determined with Sasaki's growth data in the current model.

Discussion

Although a specific cause (changes in sampling method, environmental factors, or differences in fish abundance) and time of the changes observed in sablefish growth were not identified, these changes have occurred. The division of the sablefish age-and-length data set into 2 growth regimes was not based on any detectable shift in growth but on a change in the sampling design of the longline survey. Separating the data into 2 time intervals might not completely capture the temporal pattern of changes in growth. Sablefish growth, for example, might have changed slowly, instead of in a stepwise fashion. However, we also analyzed growth data by individual years, and no obvious temporal patterns were noted. It is unlikely that other changes in the survey could explain apparent differences in growth over the time series because of the standardization in most other aspects of the survey design between the 2 periods (Hanselman et al., 2007). In addition, both the ages and the abundance indices for the sablefish stock assessment are treated as separate surveys (Japan-U.S. cooperative and domestic NMFS longline surveys) with different catchability values and sensitivities, and therefore updated growth estimates fitted to these 2 survey periods (which theoretically represent the 2 surveys) follow accordingly.

Aging error could be a very plausible cause for seeing changes in growth when in fact growth has not changed. Although Heifetz et al. (1999), among others, have validated the currently accepted aging practices (Beamish and Chilton, 1982) and have examined sources of error in the aging of sablefish, there is still much disagreement on the possibility of obtaining reliable ages from sablefish otoliths (Pearson and Shaw, 2004). We feel aging error is an unlikely cause for the growth changes seen in this study. The NMFS stock assessment of sablefish in Alaska uses an aging error matrix, one with known ages that make it particularly realistic. Aging error should not have a major effect on growth estimation if the aging error is imprecise but not biased, and Heifetz et al. (1999) found bias to be at a minimum for younger ages when most growth is occurring. In addition, otoliths for Alaskan sablefish have been read consistently by the same age-reader using the same protocol during the timeline of this study and age-reader agreement tests have been in place throughout that entire time thus, removing the possibility of an age-reader effect.

The best documented causes for change in growth of various fish species (juvenile sablefish; Pacific halibut; yellowtail flounder [*Limanda ferruginea*]; haddock

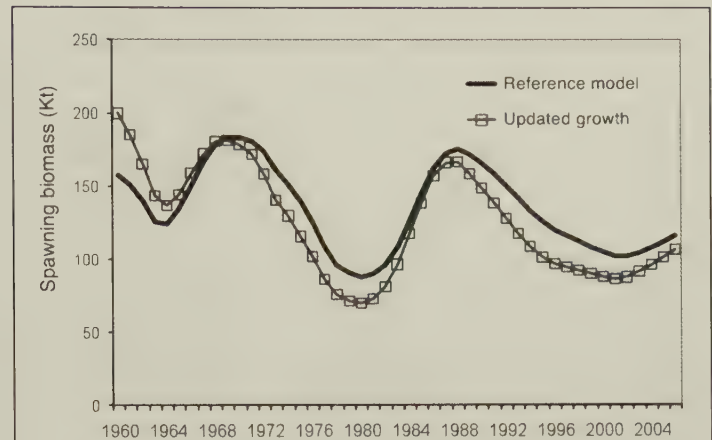


Figure 5

Comparison of time series of spawning biomass, measured in kilotons (kt), of female sablefish (*Anoplopoma fimbria*) in Alaska from the sablefish reference model (Hanselman et al., 2006; shown as a solid line) and from the same model with updated growth parameters from our study (shown as line with open boxes).

[*Melanogrammus aeglefinus*]; and Pacific chub mackerel [*Scomber japonicas*] have been density dependence and environmental conditions (Ross and Nelson, 1992; Clark et al., 1999; Wilson, 2000; Sogard and Olla, 2001; Watanabe and Yatsu, 2004). In our study, we did find evidence that changes in growth may be the result of intraspecific density-dependent mechanisms. It appears that sablefish growth is most influenced by fish density that fish are exposed to while in the larval and juvenile stages. This response in turn is linked highly to favorable environmental conditions for recruitment and YOY survival (McFarlane and Beamish, 1992; Sigler and Lunsford, 2001; Sogard and Olla, 2001). Results of our growth analysis show that sablefish from the more recent time period of our study (1996–2004), when compared to sablefish from earlier time period of our study (1981–93), exhibited faster growth rates and reached larger sizes-at-age as biomass steadily declined (Hanselman et al., 2007). Although the Alaskan sablefish population is considered to be at a sustainable, healthy level and is neither overfished nor approaching an overfished level, it is by no means close to its peak abundances of the late 1980s and early 1990s (Hanselman et al., 2010). Across the time series, abundance of Alaskan sablefish was characterized by relatively consistent high values (e.g., age-4+ abundance of 489 kt in 1986) during the early period of this study and consistently lower values (e.g., age-4+ abundance of 223 kt in 2000) during the more recent period (Hanselman et al., 2010). Since 1988, abundance has decreased substantially, whereas growth has increased significantly (Hanselman et al., 2007).

Although no direct relationships were observed between sablefish growth and any of the tested environmental factors, it is important to note that evaluating

which environmental variables are appropriate proxies for the ambient conditions that influence growth may be done best with data from smaller time and space scales than with the data available for the purposes of this study and that environmental data at fine temporal or spatial scales are likely to be difficult to interpret for fish species that move long distances (Heifetz and Fujioka, 1991; Kimura et al., 1998). The use of broad geographic and time-averaged representations of environmental effects misses short-term changes in temperature regimes brought on by weather events, such as wind-driven mixing and upwelling. In the future, to determine an appropriate scale, results from the extensive tagging studies with sablefish should be examined (the Fisheries Agency of Japan and NMFS have been tagging sablefish throughout the entire geographic range of the annual longline survey since 1972). Further, analysis should be done with the El Niño-Southern Oscillation as an environmental variable in a similar manner to work done by Kimura et al. (1998) who found growth of groundfish species to be significantly enhanced by events of the El Niño-Southern Oscillation.

There appear to be significant differences in growth patterns among management regions; the GOA regions consistently displayed the largest (in asymptotic length and average size) sablefish for both sexes in this study and in past research (McDevitt, 1990; Sigler et al., 1997). Sasaki (1985) reported regional differences in mean sizes between young sablefish from the EBS, AI, and GOA and a temporal increase in weight-at-age in the EBS from the 1960s to the late 1970s similar to the temporal increase in growth (length-at-age) reported here. Sasaki's reported differences were minor and not significant. McDevitt (1990) reported significant growth differences between the EBS and GOA but did not find significant differences in growth between the AI and EBS and the AI and GOA. She speculated that her findings were the result of high variability of the data from the AI. Consequently, differences between the AI and EBS regions were not detected because of the low power of the tests. In accord with our results, Sigler et al. (1997) found that female sablefish in the Shumagin and Southeast regions of the GOA differed significantly in growth, but no regional differences were detected for males.

In both the AI and EBS regions, poor model fits and atypical rates of growth and average maximum sizes were noted in this and past studies (McDevitt, 1990). Both of these regions displayed notably high estimates of the growth parameter k , likely because samples from these two regions consisted mostly of larger (>66 cm FL) fish, and smaller (<57 cm FL) fish are required for an accurate estimate of k . Data from both of these regions exhibit the highest variability (large residual population variances) and the poorest fit to the growth curves, compared with data from other regions in this study. The most notable differences among observed sablefish were consistently found in the EBS region, where smaller asymptotic lengths were reported than those for sablefish found in all other regions.

Alaskan sablefish are assessed by the AFSC as one stock, and therefore sablefish found throughout Alaskan waters are assumed to display similar growth rates; however, this stock is highly mobile (Heifetz and Fujioka, 1991; Maloney and Sigler, 2008). Younger fish move into deeper waters onto the slope, moving from the Eastern Gulf of Alaska (EGOA) in a counter clockwise direction through the Central Gulf of Alaska (CGOA) to the Western Gulf of Alaska (WGOA), returning to the EGOA as larger, older fish (Heifetz and Fujioka, 1991; Maloney and Sigler, 2008). In theory, one would not expect there to be many regional differences in sablefish growth and average size-at-age because a large part of the sablefish population moves each year, maintaining a well-mixed population (Heifetz and Fujioka, 1991). Several competing hypotheses are available to explain these observed regional differences: geographic differences in food abundance, oceanographic condition, or sablefish abundance. Any explanation for these regional differences, however, has to be consistent with this observed movement pattern. As with observed geographical variation for the northern anchovy (*Engraulis mordax*) along the west coast of North America, geographical variation in age composition could have influenced the observed variation in mean size of sablefish in the 6 management regions (Parrish et al., 1985; Saunders et al., 1997). Sablefish in the GOA may have displayed apparently faster growth and larger asymptotic lengths and weights than have sablefish in the AI and EBS regions because size-dependent migration results in a mixture of faster-growing young fish with older spawning fish (Heifetz and Fujioka, 1991). In contrast, in the EBS region, which primarily comprises fish >4 years of age, sablefish might have displayed slower growth because of the absence of the youngest, fastest-growing fish (Quinn and Deriso, 1999; Sogard and Olla, 2001).

Alternatively, varying growth rates might be explained in part by regional differences in abiotic factors, such as oceanographic conditions (Sasaki, 1985; McDevitt, 1990; Saunders et al., 1997; Kuznetsova, 2003). If fish are grouped within a highly migratory population, environmental effects would appear as growth differences between the 6 management regions. Temperature differences may explain the divergence in growth rates between fish in the EBS region and fish in regions in the GOA, such as the Southeast region. Several marine species (e.g., northern anchovy; Atlantic cod [*Gadus morhua* L.]; walleye pollock [*Theragra chalcogramma*]; turbot [*Scophthalmus maximus*]; and blacknose shark [*Carcharhinus acronotus*]) are of larger sizes and are faster growing in the southern extent of their ranges than in other areas of their distributions (Parrish et al., 1985; Imsland et al., 2001; Kuznetsova, 2003; Armstong et al., 2004; Driggers et al., 2004; Stark et al., 2007).

For the purposes of the management of Alaskan sablefish, updated and corrected growth estimates divided into the 2 survey periods, 1981–93 and 1996–2004, have been incorporated into the Alaskan sablefish stock assessment conducted by the AFSC. We ran the model

used in our study with the data updated through 2010 and found results that were not significantly different from the results of our analysis with data collected through 2004. Therefore, our initial results were used: VB parameter estimates for females from 1996–2010, $L_{\infty}=79.9$, $k=0.22$, $t_0=-2.23$; VB parameter estimates for males from 1996–2010, $L_{\infty}=68$, $k=0.273$, $t_0=-3.01$. The updated growth estimates provide a better fit to the data, and they are the result of decades more age and growth collections with previous size biases corrected. We view these updated growth estimates as a needed and substantial increase in biological realism for the Alaskan sablefish stock assessment model. In the future, growth will be revisited periodically, but as data accumulate, the addition of the newest data should have only nominal effects on recommendations for harvest rates (Hanselman et al., 2007).

Conclusions

In moving closer to estimating true underlying sablefish growth, we have revealed that, historically, the sizes of sablefish modeled in the Alaskan sablefish stock assessment were slightly too large. This study aids in describing the population of sablefish in Alaska more realistically as having a smaller maximum size. The use of these improved estimates will result in more conservative management in the short term but more harvest stability in the future. Although a specific cause and time for the changes in sablefish growth was not identified, these changes have occurred. To properly manage this important economic resource, the updated estimates for growth should continue to be used for the NMFS assessment of the Alaskan sablefish stock.

This study provides an example of the importance of identifying and correcting for biases that may be produced from different data collection strategies and of scientists periodically revisiting life history parameter estimates used in assessment of various stocks. The result of such efforts could mean the difference in overestimation or underestimation of abundance and, in turn, could have an effect on allotted harvest recommendations.

Acknowledgments

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The award for best publication of the year is given to authors who are employees of the National Marine Fisheries Service and whose article is judged to be the most noteworthy of those published in *Fishery Bulletin* and *Marine Fisheries Review*. Authors from the National Marine Fisheries Service are noted in bold font below.

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Ralston, Stephen, André E. Punt, **Owen S. Hamel**, John D. DeVore, and **Ramon J. Conser**

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The winners for *Marine Fisheries Review*:

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Soviet illegal whaling: the devil and the details
Mar. Fish. Rev. 73(3):1–19.

Errata

Fishery Bulletin 110(2):156–175.

Rudershausen, Paul J., Jeffrey A. Buckel, Greg E. Bolton,
Randy W. Gregory, Tyler W. Averett, and Paul B. Conn
A comparison between circle hook and J hook performance
in the dolphinfish, yellowfin tuna, and wahoo troll fishery
off North Carolina

Corrections

The title of this article should read as follows:

A comparison between circle hook and J hook performance
in the dolphinfish, yellowfin tuna, and wahoo troll fishery off
the coast of North Carolina

For the general text and figure and table legends:

The correct species name for skipjack tuna is *Katsuwonus*
pelamis and the correct common name for *Euthynnus*
alletteratus is “little tunny.”

Fishery Bulletin

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